7. Injuries to Wildlife

This chapter presents the Stage I Injury Assessment for wildlife resources of the Kalamazoo River and Portage Creek. The majority of the Kalamazoo River corridor downstream of the city of Kalamazoo is relatively undeveloped. The lower Kalamazoo River is designated a "Natural River" under authority of Michigan's Natural Rivers Act (Part 305 P.A. 451, 1994) (MDNR, 2002). This program is in place to preserve, protect, and enhance Michigan's river systems by maintaining these rivers and adjoining land in as natural a state as possible by preventing unwise use and development. The river's adjoining riparian areas provide essential habitat for many wildlife species because of proximity to the water. Wildlife frequently use travel corridors where vegetation extends along the river and its banks for much of its length (MDNR, 2002).

The Kalamazoo River contains some of the most extensive riparian habitat in southwestern Michigan, providing ample habitat for wildlife (Blasland, Bouck & Lee, 2000c). Sections of the Kalamazoo River corridor, including the Allegan State Game Area and the private Pottawattamie Fish and Game Club, are reserved and managed specifically for wildlife resources. Riparian zones along the Kalamazoo River provide food and cover for both aquatic organisms and terrestrial organisms (Blasland, Bouck & Lee, 2000c).

Some of the riparian areas of the KRE contain high concentrations of PCBs deposited by the river as well as habitat for wildlife. The former impoundments behind the Plainwell, Otsego, and Trowbridge dams contain approximately 510 acres of former sediments as floodplain soils, much of which contains relatively high concentrations of PCBs (Blasland, Bouck & Lee, 1992, 2000b). These former impoundment areas also support wildlife habitat, creating the potential for wildlife to be exposed to PCBs in these areas. For example, the former Trowbridge impoundment contains 374 acres of palustrine forested, emergent, and scrub/shrub wetlands (Figure 7.1).

The riparian wetland habitat of the Kalamazoo River supports many birds, including waterfowl, game birds, raptors, and songbirds (Blasland, Bouck & Lee, 2000c). Extensive marshes, especially downstream of Lake Allegan, provide important resting and feeding habitat for waterfowl, shorebirds, and other birds during migration. Bird surveys were conducted in riparian habitat along the river from Battle Creek to Saugatuck in the spring of 1992 and 1993 by the Kalamazoo River Nature Center, and transects in the Allegan State Game area were resurveyed in 1994 (Adams et al., 1998). Approximately 100 species were observed during each year of the survey (Adams et al., 1998). Species observed in these surveys that use wetland habitat are listed in Table 7.1. A high proportion (about 60%) of birds observed along the Kalamazoo River are neotropical migrants, which breed in the United States or Canada and migrate to Central or South America for winter. Other species use the Kalamazoo River area as winter habitat. Resident species are also present (Adams et al., 1998). It should be noted that the survey methods favored

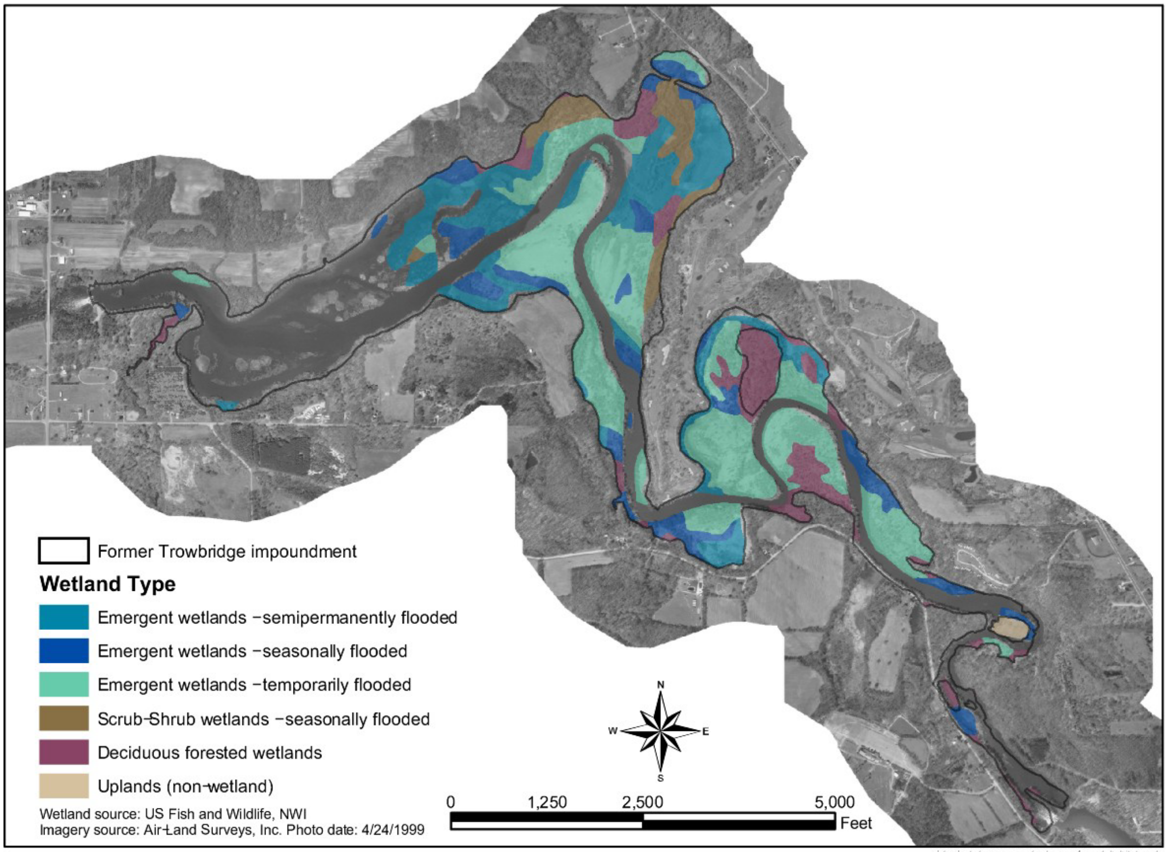

Figure 7.1. Wetland types within the former Trowbridge impoundment.

the observation of birds which were active in more open areas (Adams et al., 1998). Many species of birds in addition to those presented in Table 7.1 are expected to be found in the Kalamazoo River Basin (Blasland, Bouck & Lee, 2000c).

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status

Scientific name	Common name	Status[a]
Birds		
Podilymbus podiceps	Pied-billed grebe	
Ardea herodias	Great blue heron	
Ardea alba	Great egret	
Butorides virescens	Green heron	
Nycticorax nycticorax	Black-crowned night-heron	MI - SC
Cathartes aura	Turkey vulture	
Branta canadensis	Canada goose	
Cygnus olor	Mute swan	
Aix sponsa	Wood duck	
Anas rubripes	American black duck	
Anas platyrhynchos	Mallard	
Anas discors	Blue-winged teal	
Bucephala clangula	Common goldeneye	
Lophodytes cucullatus	Hooded merganser	
Pandion haliaetus	Osprey	MI - T
Haliaeetus leucocephalus	Bald eagle	US - T
Circus cyaneus	Northern harrier	MI - SC
Accipiter striatus	Sharp-shinned hawk	
Buteo jamaicensis	Red-tailed hawk	
Falco sparverius	American kestrel	
Phasianus colchicus	Ring-necked pheasant	
Bonasa umbellus	Ruffed grouse	
Meleagris gallopavo	Wild turkey	
Grus canadensis	Sandhill crane	
Charadrius vociferus	Killdeer	
Actitis macularia	Spotted sandpiper	
Scolopax minor	American woodcock	
Larus delawarensis	Ring-billed gull	
Chlidonias niger	Black tern	MI - SC
Columba livia	Rock dove	
Zenaida macroura	Mourning dove	
Coccyzus erythropthalmus	Black-billed cuckoo	
Coccyzus americanus	Yellow-billed cuckoo	
Otus asio	Eastern screech-owl	

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status (cont.)

Scientific name	Common name	Status[a]
Bubo virginianus	Great horned owl	
Strix varia	Barred owl	
Chaetura pelagica	Chimney swift	
Archilochus colubris	Ruby-throated hummingbird	
Ceryle alcyon	Belted kingfisher	
Melanerpes erythrocephalus	Red-headed woodpecker	
Melanerpes carolinus	Red-bellied woodpecker	
Sphyrapicus varius	Yellow-bellied sapsucker	
Picoides pubescens	Downy woodpecker	
Picoides villosus	Hairy woodpecker	
Colaptes auratus	Northern flicker	
Dryocopus pileatus	Pileated woodpecker	
Contopus virens	Eastern wood-pewee	
Empidonax virescens	Acadian flycatcher	
Empidonax traillii	Willow flycatcher	
Empidonax minimus	Least flycatcher	
Sayornis phoebe	Eastern phoebe	
Myiarchus crinitus	Great crested flycatcher	
Tyrannus tyrannus	Eastern kingbird	
Vireo flavifrons	Yellow-throated vireo	
Vireo solitarius	Blue-headed (solitary) vireo	
Vireo gilvus	Warbling vireo	
Vireo olivaceus	Red-eyed vireo	
Cyanocitta cristata	Blue jay	
Corvus brachyrhynchos	American crow	
Progne subis	Purple martin	
Tachycineta bicolor	Tree swallow	
Stelgidopteryx serripennis	Northern rough-winged swallow	
Riparia riparia	Bank swallow	
Petrochelidon pyrrhonota	Cliff swallow	
Hirundo rustica	Barn swallow	
Poecile atricapilla	Black-capped chickadee	
Baeolophus bicolor	Tufted titmouse	
Sitta carolinensis	White-breasted nuthatch	
Certhia americana	Brown creeper	

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status (cont.)

Scientific name	Common name	Status[a]
Thryothorus ludovicianus	Carolina wren	
Troglodytes aedon	House wren	
Cistothorus palustris	Marsh wren	MI - SC
Polioptila caerulea	Blue-gray gnatcatcher	
Sialia sialis	Eastern bluebird	
Catharus fuscescens	Veery	
Hylocichla mustelina	Wood thrush	
Turdus migratorius	American robin	
Dumetella carolinensis	Gray catbird	
Toxostoma rufum	Brown thrasher	
Sturnus vulgaris	European starling	
Bombycilla cedrorum	Cedar waxwing	
Vermivora pinus	Blue-winged warbler	
Vermivora peregrina	Tennessee warbler	
Parula americana	Northern parula warbler	
Dendroica petechia	Yellow warbler	
Dendroica pensylvanica	Chestnut-sided warbler	
Dendroica caerulescens	Black-throated blue warbler	
Dendroica virens	Black-throated green warbler	
Dendroica fusca	Blackburnian warbler	
Dendroica cerulea	Cerulean warbler	MI - SC
Setophaga ruticilla	American redstart	
Protonotaria citrea	Prothonotary warbler	MI - SC
Seiurus aurocapillus	Ovenbird	
Seiurus motacilla	Louisiana waterthrush	MI - SC
Oporornis philadelphia	Mourning warbler	
Geothlypis trichas	Common yellowthroat	
Piranga olivacea	Scarlet tanager	
Pipilo erythrophthalmus	Eastern towhee	
Spizella passerina	Chipping sparrow	
Spizella pusilla	Field sparrow	
Melospiza melodia	Song sparrow	
Melospiza georgiana	Swamp sparrow	
Cardinalis cardinalis	Northern cardinal	
Pheucticus ludovicianus	Rose-breasted grosbeak	

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status (cont.)

Scientific name	Common name	Status[a]
Passerina cyanea	Indigo bunting	
Agelaius phoeniceus	Red-winged blackbird	
Sturnella magna	Eastern meadowlark	
Quiscalus quiscula	Common grackle	
Molothrus ater	Brown-headed cowbird	
Icterus galbula	Baltimore oriole	
Carpodacus mexicanus	House finch	
Carduelis tristis	American goldfinch	
Coccothraustes vespertinus	Evening grosbeak	
Passer domesticus	House sparrow	
Amphibians		
Acris crepitans blanchardi	Blanchard's cricket frog	MI - SC
Ambystoma laterale	Blue-spotted salamander	
Ambystoma maculatum	Spotted salamander	
Ambystoma opacum	Marbled salamander	MI - T
Ambystoma tigrinum	Tiger salamander	
Bufo americanus	American toad	
Bufo fowleri	Fowler's toad	
Hemidactylium scutatum	Four-toed salamander	
Hyla versicolor	Gray treefrog	
Necturus maculosus	Mudpuppy	
Notophthalmus viridescens	Eastern newt	
Plethodon cinereus	Eastern red-backed salamander	
Pseudacris crucifer	Spring peeper	
Pseudacris triseriata	Western chorus frog	
Rana catesbeiana	American bullfrog	
Rana clamitans	Green frog	
Rana palustris	Pickerel frog	
Rana pipiens	Northern leopard frog	
Rana sylvatica	Wood frog	
Reptiles		
Apalone spinifera	Eastern spiny softshell	
Chelydra serpentina	Snapping turtle	
Chrysemys picta	Painted turtle	
Clemmys guttata	Spotted turtle	MI - T

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status (cont.)

Scientific name	Common name	Status[a]
Clonophis kirtlandii	Kirtland's snake	MI - E
Coluber constrictor foxii	Blue racer	
Diadophis punctatus edwardi	Northern ringneck snake	
Elaphe obsoleta obsoleta	Black rat snake	MI - SC
Emydoidea blandingii	Blanding's turtle	MI - SC
Graptemys geographica	Map turtle	
Eumeces fasciatus	Five-lined skink	
Heterodon platirhinos	Eastern hognose snake	
Lampropeltis trianguium traingulum	Eastern milk snake	
Nerodia sipedon sipedon	Northern water snake	
Opheodrys vernalis	Smooth green snake	
Regina septemvittata	Queen snake	
Sistrurus catenatus catenatus	Eastern massasauga rattlesnake	MI - SC; US- C
Sternotherus odoratus	Musk turtle (stinkpot)	
Storeria dekayi	Brown snake	
Storeria occipitomaculata occipitomaculata	Northern red-bellied snake	
Terrapene carolina carolina	Eastern box turtle	MI - SC
Thamnophis butleri	Butler's garter snake	
Thamnophis sauritus septentrionalis	Northern ribbon snake	
Thamnophis sirtalis sirtalis	Eastern garter snake	
Mammals		
Blarina brevicauda	Shorttail shrew	
Canis latrans	Coyote	
Castor canadensis	Beaver	
Condylura cristata	Starnose mole	
Cryptotis parva	Least shrew	MI - T
Didelphis marsupialis	Opossum	
Eptesicus fuscus	Big brown bat	
Erethizon dorsatum	Porcupine	
Felis rufus	Bobcat	
Glaucomys volans	Southern flying squirrel	
Lasionycteris noctivagans	Silver-haired bat	
Lasiurus borealis	Red bat	
Lasiurus cinereus	Hoary bat	

Table 7.1. Wildlife species observed in the Kalamazoo River Basin that utilize wetland habitat, and their protection status (cont.)

Scientific name	Common name	Status[a]
Lutra canadensis	River otter	
Marmota monax	Woodchuck	
Mephitis mephitis	Striped skunk	
Microtus pinetorum	Woodland vole	MI - SC
Microtus ochrogaster	Prairie vole	MI - E
Microtus pennsylvanicus	Meadow vole	
Mus musculus	House mouse	
Mustela erminea	Ermine	
Mustela frenata	Longtail weasel	
Mustela nivalis	Least weasel	
Mustela vison	Mink	
Myotis keenii	Keen’s bat	
Myotis lucifugus	Little brown bat	
Nycticeius humeralis	Evening bat	
Odocoileus virginianus	Whitetail deer	
Ondatra zibethicus	Muskrat	
Peromyscus leucopus	White-footed mouse	
Peromyscus maniculatus	Deer mouse	
Procyon lotor	Raccoon	
Scalopus aquaticus	Eastern mole	
Sciurus carolinensis	Eastern gray squirrel	
Sciurus niger	Eastern fox squirrel	
Sorex cinereus	Masked shrew	
Spermophilus tridecemlineatus	Thirteen-lined ground squirrel	
Sylvilagus floridanus	Eastern cottontail	
Synaptomys cooperi	Southern bog lemming	
Tamias striatus	Eastern chipmunk	
Tamiasciurus hudsonicus	Red squirrel	
Taxidea taxus	Badger	
Urocyon cinereoargenteus	Gray fox	
Vulpes vulpes	Red fox	
Zapus hudsonius	Meadow jumping mouse	

a. State listings (MI) from Michigan Natural Features Inventory (2002); Federal (U.S.) from U.S. FWS (2003b). E = endangered, T = threatened, SC = special concern, C = under consideration for listing.

Source: Birds from Adams et al. (1998); other animals from Blasland, Bouck & Lee (2000c).

Nine avian species identified as endangered, threatened or of special concern were observed during the surveys: black-crowned night heron, black tern, bald eagle, northern harrier, osprey, marsh wren, prothonotary warbler, cerulean warbler, and Louisiana waterthrush (Adams et al., 1998; Michigan Natural Features Inventory, 2002; U.S. FWS, 2003a).

Reptilian, amphibian, and mammalian species that utilize riparian habitat in the KRE are also listed in Table 7.1. These species have been observed in the Allegan State Game Area (MDNR, 1993b, as cited in Blasland, Bouck & Lee, 2000c). Several species are identified as endangered, threatened or of special concern, including Blanchard's cricket frog, marbled salamander, spotted turtle, Kirtland's snake, black rat snake, Blanding's turtle, eastern massasauga rattlesnake, eastern box turtle, least shrew, woodland vole and prairie vole.

The Indiana bat (*Myotis sodalis*) is listed as endangered by both the state and federal governments and may be present in the KRE even though it was not reported as observed in the KRE by Blasland, Bouck & Lee (2000c). The U.S. FWS considers that Indiana bats may be present in suitable habitat throughout the southern three tiers of counties in Michigan. Suitable summer habitat for the Indiana bat consists of floodplain and upland forests with roost trees that have exfoliating bark. The KRE contains such habitat and is within the known range of the Indiana bat, so the bat may be present and detectable using species-specific survey methods.

Ecosystem services provided by wildlife include prey for carnivorous and omnivorous wildlife, control of prey populations, and nutrient and energy cycling. Human use services include various types of recreation (hunting, birdwatching) and as supplemental food sources.

7.1 Injury Definitions

Biological resources are defined in the DOI regulations as "those natural resources referred to in section 101(16) of CERCLA as fish and wildlife and other biota. Fish and wildlife include marine and freshwater aquatic and terrestrial species; game, nongame, and commercial species; and threatened, endangered, and state sensitive species. Other biota encompass shellfish, terrestrial and aquatic plants, and other living organisms not listed in this definition" [43 C.F.R. § 11.14(f)]. This chapter addresses injuries to wildlife; injuries to aquatic biota were addressed separately in Chapters 5 and 6 of this document.

According to DOI regulations, "an injury to a biological resource has resulted from the . . . release of a hazardous substance if concentration of the substance is sufficient to" [43 C.F.R. § 11.62(f)(1)]:

- Exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 342, in edible portions of organisms [43 C.F.R. § 11.62 (f)(1)(ii)]

- Cause the biological resource or its offspring to have undergone at least one of the following adverse changes in viability: death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions (including malfunctions in reproduction), or physical deformations [43 C.F.R. § 11.62(f)(1)(i)].

An injury to biological resources can be demonstrated, per the DOI regulations, "if the biological response under consideration can satisfy all of the following acceptance criteria" [43 C.F.R. § 11.62 (f)(2)]: (i-iv)]:

- The biological response is often the result of exposure to . . . [the] hazardous substances [43 C.F.R. § 11.62 (f)(2)(i)].

- Exposure to . . . [the] hazardous substances is known to cause this biological response in free-ranging organisms [43 C.F.R. § 11.62 (f)(2)(ii)].

- Exposure to . . . [the] hazardous substances is known to cause this biological response in controlled experiments [43 C.F.R. § 11.62 (f)(2)(iii)].

- The biological response measurement is practical to perform and produces scientifically valid results [43 C.F.R. § 11.62 (f)(2)(iv)].

Injuries to biological resources may include death [43 C.F.R. § 11.62 (f)(4)(i)], as confirmed by laboratory toxicity testing [43 C.F.R. § 11.62 (f)(4)(i)(E)], behavioral abnormalities [43 C.F.R. § 11.62 (f)(4)(iii)(B)], and physiological malfunctions [43 C.F.R. § 11.62 (f)(4)(v)].

7.2 Stage I Injury Assessment Approach

Exposure to PCBs is known to cause a wide range of adverse effects in birds (outlined in Table 7.2). PCB exposure can cause death in avian embryos and juvenile and adult birds as well as sublethal and reproductive effects. Female reproductive systems in birds are generally the most sensitive endpoints to PCB toxicity (Peterson et al., 1993; Safe, 1994). PCBs have been found to cause chromosome alteration and to increase susceptibility to disease. Behavioral abnormalities such as impaired courtship, abnormal nest building behavior, and impaired avoidance behavior have also been observed in birds exposed to PCBs. Physiological impairments caused by PCB exposure in birds include decreased fecundity and altered biochemistry. Physical deformations such as beak and skeletal deformities, increased liver weights, and histopathological lesions are also known effects of PCB exposure in avian species.

Table 7.2. Overview of adverse effects in birds caused by exposure to PCBs

Category	Response measure	Documented response	Example studies
Death	Mortality	Increased adult and juvenile mortality	Heath et al., 1972; Stickel et al., 1984
		Increased embryomortality	Carlson and Duby, 1973; Brunström and Reutergårdh, 1986
Cancer/genetic mutation	Genetic mutation	Chromosome alteration	Peakall et al., 1972
Disease	Immune system impairment	Increased susceptibility to viral challenge	Friend and Trainer, 1970
Behavioral abnormalities	Reproductive behavior impairment	Reduced parental incubation attentiveness	Peakall and Peakall, 1973
		Impaired courtship behavior	Tori and Peterle, 1983
Behavioral abnormalities	Behavioral impairment	Abnormal nest building behavior	McCarty and Secord, 1999
		Impaired avoidance of visual cliff	Dahlgren and Linder, 1971
Physiological malfunction	Reproductive impairment	Reduced fecundity	Lincer and Peakall, 1970; Peakall et al., 1972; Carlson and Duby, 1973; Brunström and Reutergårdh, 1986
		Eggshell thinning	Haseltine and Prouty, 1980
Physiological malfunction	Biochemical changes	Reduced estrogen levels	Chen et al., 1994
		Porphyria	Elliott et al., 1990
		Altered vitamin A status	Spear et al., 1989
		Enzyme induction	Brunström and Lund, 1988
Physical deformation	Deformities	External malformation (e.g., small beak, eyes, unabsorbed yolk sac)	Brunström and Lund, 1988; Hoffman et al., 1998
		Skeletal deformities	Hoffman et al., 1998
		Increased heart weights	Hansen et al., 1976; Heid et al., 2001
		Increased liver weights	Elliott et al., 1997
		Histopathological liver lesions	Hoffman et al., 1996b

Mammalian exposure to PCBs has been found to cause similar types of effects (Table 7.3). As with birds, female reproductive systems are generally the most sensitive endpoints to PCB toxicity. Exposure to PCBs can result in mortality, sublethal effects, and reproductive impairment in mammals. Behavioral abnormalities, such as learning deficits, have been observed in rats exposed to PCBs via maternal exposure. Physiological impairment effects include reduced reproductive success and offspring mortality as well as altered vitamin status and enzyme induction. PCB exposure has also been known to cause physical deformities such as jaw deformities, increased organ weights, and deformed nail growth in mink. Mink are particularly sensitive to PCBs and have been studied extensively.

Table 7.3. Overview of adverse effects in mammals caused by exposure to PCBs

Category	Response measure	Documented response	Example studies
Death	Mortality	Increased mortality	Aulerich et al., 1973, 1985; Platonow and Karstad, 1973; Aulerich and Ringer, 1977; Bleavins et al., 1980
Behavioral abnormalities	Neurological impairment	Neurodevelopmental deficits	Lilienthal and Winneke, 1991
Behavioral abnormalities	Dietary impairment	Refusal to eat/anorexia	Aulerich et al., 1985, 1987; Ringer, 1983
Behavioral abnormalities	Reproductive impairment	Decreased sexual receptivity	Brezner et al., 1984
		Delayed copulation	Brezner et al., 1984
Physiological malfunction	Reproductive impairment	Reduced reproductive success	Platonow and Karstad, 1973; Aulerich et al., 1985; Bäcklin and Bergman, 1992; Restum et al., 1998
		Mortality of offspring	Hornshaw et al., 1983; Ringer, 1983; Brezner et al., 1984; Aulerich et al., 1985; Wren et al., 1987b; Heaton et al., 1995a; Restum et al., 1998
		Reduced weight gain in offspring	Brezner et al., 1984; Wren et al., 1987b
Physiological malfunction	Biochemical changes	Altered vitamin A status	Brunström et al., 1991; Håkansson et al., 1992; Käkelä et al., 1999
		Enzyme induction	Aulerich et al., 1985; Brunström et al., 1991; Dragnev et al., 1994; Shipp et al., 1998
		Immunosuppression	Brunström et al., 1991

Table 7.3. Overview of adverse effects in mammals caused by exposure to PCBs (cont.)

Category	Response measure	Documented response	Example studies
Physical deformation	Deformities	Increased organ weights	Hornshaw et al., 1983; Ringer, 1983; Aulerich et al., 1985, 1987; Kihlström et al., 1992; Heaton et al., 1995b; Restum et al., 1998
		Decreased heart weight	Aulerich et al., 1985
		Deformed nail growth	Bleavins et al., 1982; Aulerich et al., 1987
		Jaw deformities (loose teeth, bone loss)	Render et al., 2000
		Weight loss	Brezner et al., 1984; Aulerich et al., 1985, 1987
		Abnormal molting	Aulerich et al., 1987
		Gastric ulcers/internal bleeding	Aulerich et al., 1985, 1987

Little is known about the impact of PCBs and other contaminants on amphibians and reptiles as most studies have focused on reporting contaminant residues, rather than evaluating toxicity (Portelli and Bishop, 2000; Glennemeier and Begnoche, 2002). Bishop et al. (1991) report decreased hatching rates and increased deformity rates in snapping turtle (*Chelydra serpentina*) eggs collected from locations along Lake Ontario which are contaminated with PCBs, dioxins, furans, and pesticides, relative to eggs collected from a control site. Laboratory results indicate that hatching success of amphibians is negatively affected by PCB exposure (Glennemeier and Begnoche, 2002). Field observations in wetlands along the Kalamazoo River showed that densities of amphibian larvae and adults decreased with increased total PCBs in sediments, although this trend was not statistically significant (Glennemeier and Begnoche, 2002). PCB concentrations measured in green frogs (*Rana clamitans*) collected from the Kalamazoo River suggest that amphibians do not concentrate PCBs as highly as other taxa like birds or fish (Glennemeier and Begnoche, 2002). Because the toxicological literature for reptiles and amphibians is not as complete as it is for birds and mammals, potential injury to these species is not evaluated in this Stage I Assessment.

Table 7.4 outlines the approaches taken in this chapter to assess injury to birds and mammals. Concentrations of PCBs in edible portions of mallards are compared to appropriate regulatory standards for human consumption. The results of the ERA conducted by MDEQ are reviewed and PCB concentrations in whole fish are compared to dietary toxicity thresholds for piscivorous birds and mammals. Bald eagle reproductive success is evaluated and measured PCB and dichlorodiphenyl dichloroethylene (DDE) concentrations in bald eagle eggs and plasma are

Table 7.4. Approaches to evaluate injury to wildlife

Injury definition	Stage I injury assessment approach	Chapter section
Exceed action or tolerance levels established under section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 342, in edible portions of organisms [43 C.F.R. § 11.62 (f)(1)(ii)].	Compare concentrations of total PCBs in mallard duck breast tissue to FDA action levels for consumption.	7.3
Cause the biological resource or its offspring to have undergone adverse changes in viability [43 C.F.R. § 11.62(f)(1)(i)].	Review MDEQ ERA and compare measured PCB concentrations in whole fish to bald eagle dietary toxicological benchmarks.	7.4
	Evaluate reproductive success of bald eagles in the KRE; compare measured PCB and DDE concentrations in eggs to toxicological benchmarks; compare measured concentrations in nestling plasma to concentrations in other locations in Michigan.	7.5
	Compare concentrations of measured PCBs in eggs of other birds to toxicological benchmarks.	7.6
	Review MDEQ ERA and compare measured PCB concentrations in whole fish to mammalian toxicological benchmarks.	7.7
	Evaluate mink viability by reviewing mink trapping success rates and comparing measured PCB concentrations in mink tissue to toxicological benchmarks.	7.8
	Compare measured PCB concentrations in small mammal, shrew, and muskrat tissue to toxicological benchmarks.	7.9
	Compare floodplain soil PCB concentrations to modeled soil toxicity thresholds.	7.10

compared to reproductive toxicological benchmarks. PCB concentrations in other avian species also are compared to appropriate toxicological benchmarks. Mink trapping rates are evaluated and mink, small mammal, and muskrat tissue concentrations are compared to toxicological benchmarks. Finally, the extensive available data on PCB concentrations in floodplain soils is compared to thresholds for effects to wildlife through bioaccumulation of PCBs into the food chain.

7.3 Mallard Duck Breast Tissue Concentrations

7.3.1 Data sources

PCB concentrations in breast tissue of mallard ducks collected in 1988 (U.S. FWS, 1989; Michigan Department of Public Health, 1990) are used to evaluate injury to wildlife in this section.

The U.S. FWS collected one mallard duck from along the Kalamazoo River in Allegan County, four from Pottawattamie Marsh, and one from Ottawa Marsh. For all mallards except one from Pottawattamie Marsh, both a skin-on and a skin-off breast sample were analyzed at the Mississippi State University Chemical Laboratory for total PCBs on a lipid normalized basis (Mississippi State University, 1990). One mallard from Pottawattamie Marsh only had a skin-on sample collected. A skin-on breast split sample from the mallard collected along the Kalamazoo River (1A) and a skin-off breast split sample from one of the mallards collected from Pottawattamie Marsh (4B) were sent to the Michigan Department of Public Health for a Quality Assurance evaluation (Michigan Department of Public Health, 1990). Four split samples were also analyzed by the Patuxent Wildlife Research Center, including the skin-on and -off portions of the mallard from along the Kalamazoo River (1A & 1B), and the skin-on and -off portions of one mallard from Pottawattamie Marsh (4A & 4B). Although PCB data are available for other waterfowl samples collected in the KRE (MDNR, 1987b), these data were not used in this evaluation because the samples were not of an edible portion, as the DOI regulations require [43 C.F.R. § 11.62 (f)(1)(ii)]. Furthermore, lipid data were not available for these samples for use in deriving estimated concentrations in edible tissues.

7.3.2 Regulatory criteria and standards

Waterfowl resources are injured if they contain concentrations of a hazardous substance sufficient to exceed action levels or tolerances established by the FDA [43 C.F.R. § 11.62 (f)(1)(ii)]. The FDA established a temporary tolerance level of 3 mg/kg ("fat basis") in poultry [21 C.F.R. § 109.30 (a)(3)]. For the purposes of the Stage I Assessment, the Trustees assume that fat basis is equivalent to a lipid-normalized PCB concentration. This level may also be applied to edible portions of wildlife hunted recreationally, such as ducks or geese. However it should be noted that people generally consume less game than commercial poultry, and this tolerance level may be overprotective.

7.3.3 Results

The PCB concentration in the skin-on breast sample from the mallard collected from along the Kalamazoo River in Allegan County (1A), 7.7 mg/kg on a lipid basis, was greater than the FDA tolerance level of 3 mg/kg (Table 7.5; Mississippi State University, 1990). However, the split sample of 1A analyzed by Michigan Department of Public Health (1990) had a reported PCB concentration of 1.8 mg/kg lipid, and PCBs were not detected at a limit of 1.0 mg/kg lipid in the split sample analyzed at Patuxent Wildlife Research Center (U.S. FWS, 1989). PCB concentrations were below the analytical detection limit (0.05 mg/kg ww) in the mallards collected from Pottawattamie and Ottawa Marsh (Mississippi State University, 1990). However, the Michigan Department of Public Health (1990) reported a PCB concentration of 1.2 mg/kg lipid in a skin-off split sample from Potawattamie Marsh (4B), and Patuxent Wildlife Research Center reported a PCB concentration of 0.8 mg/kg lipid in a skin-on split sample from Potawattamie Marsh (4A) (U.S. FWS, 1989). Due to the limited number of samples and the inconclusiveness of the analytical data, the Trustees cannot reach any conclusions about injury to waterfowl by exceedences of FDA tolerance levels.

Table 7.5. Total PCB concentrations in breast tissue of mallard ducks, 1988

Sample ID	Sample location	Sample type	Total PCB (mg/kg ww)	Total PCB (mg/kg lipid)	Analyzed by[a]
1A	Kalamazoo River, Allegan County	Skin-on	0.29	7.7	Mississippi State
1A-split	Kalamazoo River, Allegan County	Skin-on	0.079	1.8	MDPH
1A-split	Kalamazoo River, Allegan County	Skin-on	ND (0.049)	ND (1.0)	Patuxent
1B	Kalamazoo River, Allegan County	Skin-off	ND (0.05)	ND (2.9)	Mississippi State
1B-split	Kalamazoo River, Allegan County	Skin-off	ND (0.10)	ND (1.4)	Patuxent
4A	Pottawattamie Marsh	Skin-on	ND (0.05)	ND (0.6)	Mississippi State
4A-split	Pottawattamie Marsh	Skin-on	0.05	0.8	Patuxent
4B	Pottawattamie Marsh	Skin-off	ND (0.05)	ND (2.1)	Mississippi State
4B-split	Pottawattamie Marsh	Skin-off	0.025	1.2	MDPH
4B-split	Pottawattamie Marsh	Skin-off	ND (0.051)	ND (2.7)	Patuxent
5A	Pottawattamie Marsh	Skin-on	ND (0.05)	—	Mississippi State
5B	Pottawattamie Marsh	Skin-off	ND (0.05)	—	Mississippi State

Table 7.5. Total PCB concentrations in breast tissue of mallard ducks, 1988 (cont.)

Sample ID	Sample location	Sample type	Total PCB (mg/kg ww)	Total PCB (mg/kg lipid)	Analyzed by[a]
6A	Pottawattamie Marsh	Skin-on	ND (0.05)	—	Mississippi State
6B	Pottawattamie Marsh	Skin-off	ND (0.05)	—	Mississippi State
7A	Pottawattamie Marsh	Skin-on	ND (0.05)	—	Mississippi State
8A	Ottawa Marsh	Skin-on	ND (0.05)	—	Mississippi State
8B	Ottawa Marsh	Skin-off	ND (0.05)	—	Mississippi State

a. Mississippi State = Mississippi State University, 1990; MDPH = Michigan Department of Public Health, 1990; Patuxent = U.S. FWS, 1989.

7.4 PCBs in Bird Diet

In this section, the results of MDEQ's ecological risk assessment for PCB exposure to Kalamazoo River area birds are reviewed, and PCB concentrations measured in KRE fish are compared to dietary toxicity thresholds for piscivorous birds.

7.4.1 Data sources

The following data sources are used to evaluate injury to birds in this section:

- Bird dietary exposure model from Kalamazoo River Ecological Risk Assessment (Camp Dresser & McKee, 2003b)
- Data on PCB concentrations in whole fish collected between 1993 and 1999 by Blasland, Bouck & Lee (2001)
- Data on PCB concentrations in whole fish collected in 1999 and 2000 by the MDEQ (Camp Dresser & McKee, 2001, 2002b)
- Data on PCB concentrations in whole fish collected in 1999 by Michigan State University Aquatic Toxicology Laboratory (2002j).

As part of the RI for the Kalamazoo River site, MDEQ conducted an ecological risk assessment that characterizes the risk to wildlife, including birds, from exposure to PCBs in the KRE. MDEQ's ecological risk assessment was reviewed and commented on by EPA, the PRPs, and the public, and the final version (Camp Dresser & McKee, 2003b) incorporates those comments.

Fish represent an important component in the diet of certain bird species, and whole body fish PCB concentrations are used to assess injuries to birds. Blasland, Bouck & Lee (2001) collected 110 whole body samples of golden redhorse (*Moxostoma erythrurum*), northern hogsucker (*Hypentelium nigricans*), spotted sucker (*Minytrema melanops*), and white sucker throughout the KRE in 1993. In 1997, 15 whole body smallmouth bass samples were collected from Plainwell, Lake Allegan, and New Richmond. In 1999, 24 yearling bass were sampled from locations throughout the KRE and 11 white suckers were collected in Portage Creek. All of these samples were analyzed for total PCBs as Aroclors, and these data are used to estimate the dietary exposure of KRE piscivorous birds to PCBs.

The MDEQ collected 36 composite whole body samples of yearling smallmouth bass from eight locations in the Kalamazoo River between the city of Kalamazoo and New Richmond in 1999 and analyzed them for total PCBs (Camp Dresser & McKee, 2001). One composite channel catfish sample was also collected from New Richmond in this year. In 2000, the MDEQ collected five composite whole body samples of yearling white sucker from Portage Creek, one smallmouth bass sample from the Kalamazoo River in the city of Kalamazoo, and five smallmouth bass samples from Lake Allegan (Camp Dresser & McKee, 2002b).

Michigan State University Aquatic Toxicology Laboratory (2002j) collected whole body samples of five carp, four golden redhorse, five smallmouth bass, and one white sucker from Trowbridge in 1999. Additionally, five composite whole body samples of unspecified forage fish were collected from Trowbridge in 1999 (Michigan State University Aquatic Toxicology Laboratory, 2002j).

7.4.2 Toxicity reference value derivation

TRVs can be derived from controlled laboratory toxicity tests where PCBs are administered in bird diets. Dietary toxicity includes the direct effects of ingesting PCB contaminated diets and the effects on offspring when adults ingest PCB contaminated diets. The Trustees compiled and reviewed more than 200 documents to identify studies relevant to the evaluation of dietary effects of PCB concentrations. Studies with a measured dietary dose which were considered reliable are summarized in Tables 7.6-7.9.

Table 7.6. Sublethal effects of PCB exposure to birds

Species	Sex	Life stage	Exposure duration (days)	PCB type	LOEL mg/kg diet (dw)[a]	NOEL mg/kg diet (dw)[a]	Endpoint	Citation
American kestrel	Mixed	Adult	150	Aroclor 1254 Aroclor 1262	0.6		Increased hepatic enzyme activity-oestradiol	Lincer and Peakall, 1970
Ring dove	Unspecified	Adult	56	Aroclor 1254	6.4	0.64	Reduced dopamine concentration; reduced norepinephrine concentration	Heinz et al., 1980
Chicken	Unspecified	Chick	31.5	Aroclor 1248	22	11	Reduced percent hemoglobin	Rehfeld et al., 1972
Chicken	Unspecified	Chick	56	Aroclor 1242 Aroclor 1254	22		Increased liver weight relative to body weight	Hansen et al., 1976
Mallard	Mixed	Chick	10	Aroclor 1254	28		Reduced immunity to duck hepatitis virus (DHV)	Friend and Trainer, 1970
Chicken	Unspecified	Chick	31.5	Aroclor 1248	33	22	Reduced percent blood packed cell volume	Rehfeld et al., 1972
Chicken	Unspecified	Chick	31.5	Aroclor 1248	33	22	Edema	Rehfeld et al., 1971
Chicken	Female	Adult	5	Aroclor 1254	50[b]		Increased hepatic microsomal protein	Chen et al., 1994
Chicken	Mixed	Chick	49	Aroclor 1242	54		Increased zinc absorption	Turk and Heitman, 1976
Mallard	Male	Adult	35	Aroclor 1254	63[b]	12	Increased hepatic EROD/PROD activities; reduced T3 concentration	Fowles et al., 1997
Ring dove	Unspecified	Adult	56	Aroclor 1254	68	6.4	Increased liver weight; reduced hematocrit	Heinz et al., 1980
Chicken	Female	Adult	5	Aroclor 1254	102[b]	52	Increased liver weight relative to body weight; increased hepatic P-450; reduced concentration of estradiol; reduced total plasma calcium	Chen et al., 1994
American kestrel	Female	Adult	28	Aroclor 1254	107[b]		Increased aldrin hepoxidase	Elliott et al., 1997
American kestrel	Female	Adult	84	Aroclor 1254	107[b]		Increased hepatic EROD/APND activities	Elliott et al., 1997

Table 7.6. Sublethal effects of PCB exposure to birds (cont.)

Species	Sex	Life stage	Exposure duration (days)	PCB type	LOEL mg/kg diet (dw)[a]	NOEL mg/kg diet (dw)[a]	Endpoint	Citation
White pelican	Mixed	Adult	70	Aroclor 1254	158		Increased liver weight; increased spleen weight relative to body weight; decreased protein in blood	Greichus et al., 1975
Chicken	Mixed	Chick	49	Aroclor 1242	220	52	Increased calcium absorption	Turk and Heitman, 1976
Chicken	Unspecified	Chick	28	Aroclor 1242	220	110	Hydropericardium	McCune et al., 1962
Chicken	Male	Chick	42	Aroclor 1254	275		Reduced comb weight	Platonow and Funnell, 1971
Chicken	Male	Chick	63	Aroclor 1254	275		Reduced testes weight	Platonow and Funnell, 1971
Mallard	Male	Adult	35	Aroclor 1254	313[b]	63	Increased liver weight relative to body weight; increased hepatic P450; reduced plasma glucose concentration	Fowles et al., 1997
Mallard	Male	Adult	35	Aroclor 1254	313[b]		Increased thyroid weight relative to body weight	Fowles et al., 1997
Chicken	Unspecified	Chick	21	Aroclor 1242	440	220	Abdominal and subcutaneous edema, hydrocardium	Flick et al., 1965
Chicken	Unspecified	Chick	28	Aroclor 1242	440	220	Increased liver weight; enteritis; hemorrhage of internal organs	McCune et al., 1962
Chicken	Male	Chick	60	Aroclor 1260	440		Increased liver weight; reduced spleen weight	Vos and Koeman, 1970
Chicken	Male	Chick	60	Clophen A60	440		Hydropericardium, abdominal and subcutaneous edema; liver necrosis	Vos and Koeman, 1970
Chicken	Unspecified	Chick	28	Aroclor 1242	880	440	Enlarged heart; kidney and liver damage	McCune et al., 1962

a. Where not specified in source document, reported dose was assumed to be on a wet weight basis and was converted to a dry weight basis assuming 10% moisture in commercial feed (see text).
b. Extrapolated diet concentration based on body sizes and feeding rates in the study, or based on sizes and feeding rates in Sample et al. (1996).

Table 7.7. Effects of PCB exposure on egg or sperm production

Species	Sex	Life stage	Study duration (days)	Exposure duration (days)	PCB type	LOEL mg/kg diet (dw)[a]	NOEL mg/kg diet (dw)[a]	Endpoint	Citation
Chicken	Mixed	Adult	273	273	Aroclor 1254	5.5		Reduced egg production	Platonow and Reinhart, 1973
Chicken	Female	Adult	112	63	Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254	22	2.2	Reduced egg production	Lillie et al., 1974
Chicken	Female	Adult	56	56	Aroclor 1248	22	11	Reduced egg production	Scott, 1977
American kestrel	Female	Adult	> 365	100	Mixtures of Aroclors	22		Reduced egg production in second generation birds	Fernie et al., 2001
American kestrel	Male	Adult	62	62	Aroclor 1254	100[b]		Reduced sperm concentration, sperm per ejaculate	Bird et al., 1983

a. Where not specified in source document, reported dose was assumed to be on a wet weight basis and was converted to a dry weight basis assuming 10% moisture in commercial feed (see text).
b. Dose was expressed in the study as wet weight in cockerel breast (33 mg/kg). Dose was converted to a dry weight basis assuming 66% moisture in cockerel breast (see text).

Table 7.8. Effects of PCB exposure on avian offspring sublethal endpoints

Species	Life stage	Study duration (days)	Experiment duration (days)	PCB type	LOEL mg/kg diet (dw)[a]	NOEL mg/kg diet (dw)[a]	Endpoint	Citation
Chicken	Chick	112	63	Aroclor 1248 Aroclor 1254	2.2		Reduced growth rate in offspring	Lillie et al., 1974
American kestrel	Egg	180	180	Aroclor 1248	9[b]		Increased shell length and width; reduced shell weight, thickness index, thickness	Lowe and Stendell, 1991
Chicken	Chick	14	14	Aroclor 1254	11		Increased vitamin E-selenium deficiency in progeny	Combs et al., 1975
Chicken	Chick	112	56	Aroclor 1242 Aroclor 1248	11	5.5	Reduced growth rate in offspring	Lillie et al., 1975
Chicken	Embryo	112	63	Aroclor 1232	22		Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Embryo	112	63	Aroclor 1242 Aroclor 1248 Aroclor 1254	22	2.2	Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Chick	112	63	Aroclor 1232 Aroclor 1242	22	2.2	Reduced growth rate in offspring	Lillie et al., 1974

a. Where not specified in source document, reported dose was assumed to be on a wet weight basis and was converted to a dry weight basis assuming 10% moisture in commercial feed (see text).
b. Dose was expressed in the study as wet weight in cockerel breast (3 mg/kg). Dose was converted to a dry weight basis assuming 66% moisture in cockerel breast (see text).

Table 7.9. Effects of adult PCB exposure on egg hatchability, young fledged, or egg fertility

Species	Life stage	Study duration (days)	Exposure duration (days)	PCB type	LOEL mg/kg diet (dw)[a]	NOEL mg/kg diet (dw)[a]	Endpoint	Citation
Chicken	Egg	273	273	Aroclor 1254	5.5		Reduced egg fertility	Platonow and Reinhart, 1973
Chicken	Egg	84	42	Aroclor 1242	11	5.5	Reduced number of eggs hatched	Britton and Huston, 1973
Chicken	Egg	112	56	Aroclor 1232 Aroclor 1242 Aroclor 1248	11	5.5	Reduced number of eggs hatched	Lillie et al., 1975
Chicken	Egg	56	56	Aroclor 1248	11	1.1	Reduced number of eggs hatched	Scott, 1977
Chicken	Egg	21	21	Aroclor 1248	11	1.1	Reduced egg hatchability	Scott et al., 1975
Ring dove	Egg	Unknown	Unknown	Aroclor 1254	11		Reduced number of eggs hatched and young fledged	Peakall et al., 1972
Chicken	Egg	42	42	Aroclor 1242	22		Reduced number of eggs hatched	Briggs and Harris, 1973
Chicken	Embryo	98	70	Aroclor 1242 Aroclor 1254	22		Reduced number of eggs hatched	Ax and Hansen, 1975
Chicken	Embryo	112	63	Aroclor 1232	22		Reduced number of eggs hatched	Cecil et al., 1974
Chicken	Embryo	112	63	Aroclor 1242 Aroclor 1248 Aroclor 1254	22	2.2	Reduced number of eggs hatched	Cecil et al., 1974
Chicken	Chick	112	63	Aroclor 1248	22	2.2	Reduced number of eggs hatched	Lillie et al., 1974
Chicken	Egg	112	63	Aroclor 1232 Aroclor 1242 Aroclor 1248 Aroclor 1254	22	2.2	Reduced number of eggs hatched	Lillie et al., 1974
Chicken	Egg	273	98	Aroclor 1254	55	5.5	Reduced number of fertile eggs hatched	Platonow and Reinhart, 1973

a. Where not specified in source document, reported dose was assumed to be on a wet weight basis and was converted to a dry weight basis assuming 10% moisture in commercial feed (see text).

In the documents reviewed, effects concentrations were presented as a dietary concentration (mg PCB/kg dw of food), as an ingested PCB mass (mg PCB/day), or as a body-size-adjusted ingested PCB mass (mg PCB/kg body weight/day). Most data were presented as dietary concentrations (mg PCB/kg dw of food), so where possible and practical, other dietary units were converted to dietary concentrations using body size and ingestion rates (e.g., kilogram food/day) presented in the document. Alternatively, when the study did not present body weight or ingestion rates, representative body sizes and ingestion rates found in Sample et al. (1996) were used. Only PCB dose concentrations expressed as mg/day or mg/kg/day for adult birds were converted to corresponding mg/kg dw of food, since food ingestion rate and body weight can vary considerably for juvenile birds over the course of a study. Most of the studies dosed the birds by adding PCBs to commercial feed mixtures, but in reporting the PCB doses some of the studies did not specify whether the dose was on a dry weight or wet weight basis. For these studies, it was assumed that the reported dose concentration was on a wet weight basis, and a value of 10% moisture content in commercial feed (Peakall and Peakall, 1973) was used to convert to an equivalent dry weight concentration. In addition, two studies (Bird et al., 1983; Lowe and Stendell, 1991) reported PCB doses as mg/kg ww in cockerel breast, the feed used. These concentrations were converted to a dry weight basis assuming 66% moisture in the breast tissue, which is the mean percent moisture reported for waterfowl breast tissue from two studies in New York State (Kim et al., 1984, 1985).

As the studies listed in Table 7.6 show, LOEL concentrations for sublethal endpoints are highly variable, and are largely dependent on the endpoint of concern and the exposure duration, as well as the species used in the study. A 150 day study found increased hepatic enzyme activities in American kestrels at a dietary exposure concentration of 0.6 mg/kg dw in diet (Lincer and Peakall, 1970), whereas a 28 day study on the same species detected enzyme changes at a dietary concentration of 107 mg/kg dw PCB (Elliott et al., 1977). At the high end of the range, enlarged hearts and organ tissue damage were observed in a 28 day study using chicken at 880 mg/kg dw (McCune et al., 1962).

LOEL concentrations for reproductive endpoints are somewhat less variable. Reductions in the number of eggs or sperm produced have been observed in studies of chicken and American kestrel at dietary doses of 5.5 to 100 mg/kg dw (Table 7.7). Sublethal effects in offspring, such as reduced growth rates, abnormalities, and nutritional deficiencies, have been observed at dietary doses of 2.2 to 22 mg/kg dw (Table 7.8). In eight chicken studies and one ring dove study, LOEL concentrations for egg hatchability ranged from 5.5 to 55 mg/kg dw (Table 7.9). In the single study with a dietary LOEL value of 55 mg/kg dw (Platonow and Reinhart, 1973), the next lowest concentration tested was ten times less, suggesting that this value may be considerably higher than a threshold concentration.

Other feeding studies have been conducted which dosed birds with field weathered PCBs (e.g., Summer et al., 1996a; 1996b). These studies fed fish collected from contaminated areas to birds, measured PCB concentrations in the diet, and evaluated a range of toxicological effects. Because these dietary doses potentially contain other contaminants, they are not used in the development of TRVs, but are presented as supporting evidence of field impacts of weathered PCB mixtures. Summer et al. (1996a) found reduced body and liver weights in adult leghorn hens dosed with a diet of fish containing 6.6 mg/kg ww [or 25 mg/kg dw, assuming a moisture content of 74% (Connolly et al., 1992)] PCB. Summer et al. (1996b) found increased embryomortality, decreased hatching rates and various deformities in embryos and chicks of the same adult leghorn hens.

Using the laboratory studies described above and presented in Tables 7.6 to 7.9, TRV ranges can be selected for comparison with PCB concentrations in Kalamazoo River bird diets. TRVs are selected using best professional judgment to capture the central tendency of data for endpoints that appear to be most sensitive to PCB toxicity. They are intended to represent concentrations at which adverse effects are likely to be observed.

For sensitive species such as chicken, embryomortality effects are likely to be observed between 10 and 20 mg/kg PCB dw in bird diet, and sublethal effects are likely to be observed between 0.6 and 2 mg/kg dw, based on the studies listed in Tables 7.6, 7.7, and 7.8. Based on an assumed moisture content of 74% in whole body fish (Connolly et al., 1992), these TRV ranges equate to 2.6 to 5.2 mg/kg for embryomortality and 0.2 to 0.5 mg/kg for sublethal effects on a wet weight basis.

However, these TRVs are based primarily on toxicological data for chickens, and the chicken has consistently been the most sensitive species tested to date for the toxicological effects of PCBs. Therefore, a different set of TRVs is developed for bird species that are less sensitive than chickens to PCB effects. Heath et al. (1972) found that acute LD_{50} concentrations varied across different bird species, including chicken, by a factor of approximately 5. Similarly, Hoffman et al. (1995) concluded that terns, cormorants, and eagles are approximately 5 times less sensitive to PCB effects than chickens. Therefore, for the purposes of this Stage I Assessment, the Trustees use TRVs for more PCB-tolerant species that are five times the TRVs based primarily on data for chickens. Thus, the TRVs for more tolerant bird species is 13 to 26 mg/kg ww for embryomortality and 1.0 to 2.5 mg/kg ww for sublethal effects.

7.4.3 Results

PCB dietary exposure of piscivorous birds

A comparison of measured whole fish PCB concentrations in the Kalamazoo River with dietary TRVs for birds shows that piscivorous birds, such as bald eagles, herons, mergansers, and kingfishers, may be exposed to dietary PCB concentrations that are greater than TRVs. Although fish do not contain PCBs within the embryomortality TRV range for average species, 52 of the 228 fish tissue samples are within or above the range of concentrations that would be expected to cause embryomortality effects in sensitive species (Figure 7.2). Nine golden redhorse collected in 1993, 4 smallmouth bass collected in 1997, and a total of 39 smallmouth bass, carp, golden redhorse, and forage fish composite samples collected in 1999 had total PCB concentrations greater than the lower end of the effects range of 2.6 mg/kg ww.

PCB concentrations in whole fish are within or greater than the TRV range for sublethal effects in piscivorous birds (Figure 7.3). Seventy percent of samples exceed the minimum TRV for more tolerant species, and 95% exceed the minimum TRV for sensitive species. The exceedences of TRVs extend throughout Portage Creek and the mainstem of the Kalamazoo River downstream of PRP facilities. PCB concentrations in whole fish tissue have been elevated since at least 1993 and have continued through at least 2000; however, it is likely that fish have been accumulating PCBs since the time of the initial releases, and also likely that concentrations were much higher at one time than those measured in 1993. Thus it is probable that piscivorous birds such as herons, mergansers, and kingfishers have been exposed to elevated PCB concentrations in their diet since the time of the initial releases.

MDEQ ecological risk assessment

Based on a dietary exposure model and measured concentrations in the KRE, the MDEQ calculated PCB exposure concentrations in the diets of bird species of concern as part of an ecological risk assessment for the RI/FS (Camp Dresser & McKee, 2003b). The avian species evaluated in the ecological risk assessment were the robin, bald eagle, and great horned owl. Using dietary no effect and low effect TRV values derived from the toxicology literature, MDEQ then used a food chain model to estimate the dietary PCB exposure of these bird species. No effect- and low effect-based hazard quotients (HQs) for these species were then calculated as the ratio of the dietary PCB exposure to the TRVs. An HQ of less than one indicates that the species is exposed to a lower concentration in the diet than the no effect or low effect TRV value, while a HQ of greater than one indicates that the species is exposed to a higher concentration. A higher HQ suggests that a species is more at risk to PCB toxicity through dietary exposure.

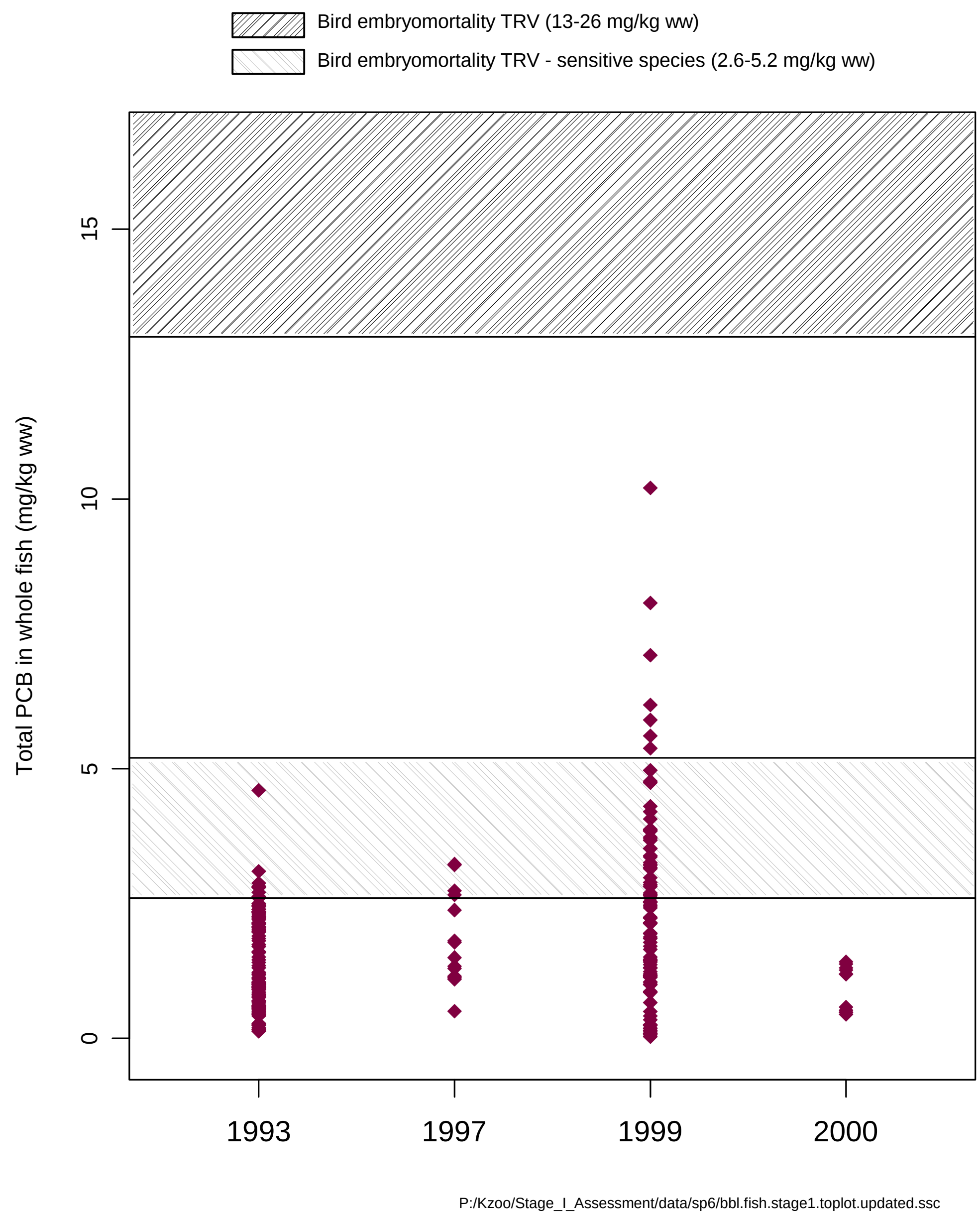

Figure 7.2. Total PCBs in whole body fish samples collected in the Kalamazoo River downstream of PRP facilities compared to dietary TRVs for embryomortality effects in birds. Species include channel catfish, golden redhorse, northern hogsucker, smallmouth bass, spotted sucker, and white sucker.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2001, 2002b; Michigan State University Aquatic Toxicology Laboratory, 2002j.

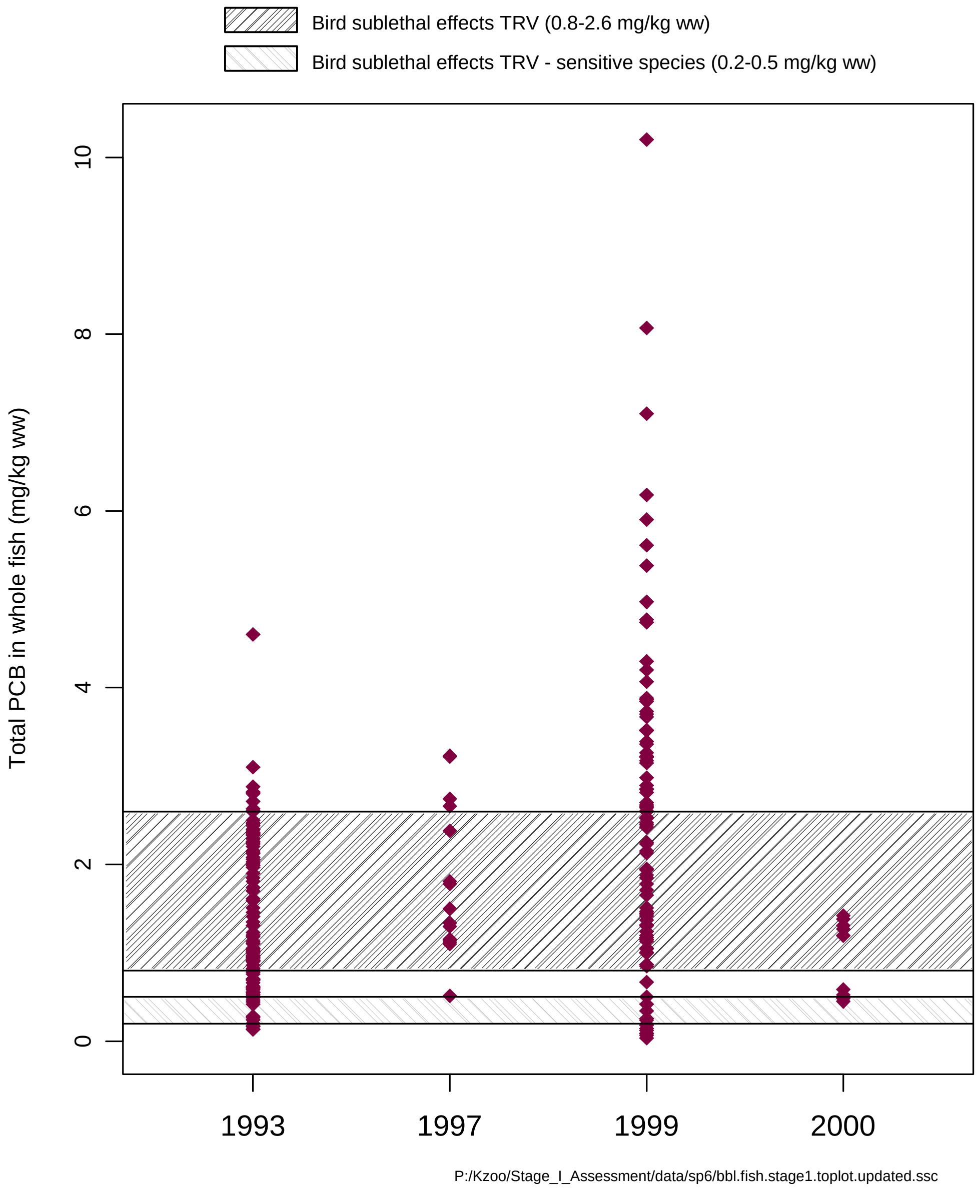

Figure 7.3. Total PCBs in whole body fish samples collected in the Kalamazoo River downstream of PRP facilities compared to dietary TRVs for sublethal effects in birds. Species include channel catfish, golden redhorse, northern hogsucker, smallmouth bass, spotted sucker, and white sucker.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2001, 2002b; Michigan State University Aquatic Toxicology Laboratory, 2002j.

MDEQ concluded that KRE avian receptors are at risk from dietary exposure to PCBs (Table 7.10; Camp Dresser & McKee, 2003b). Piscivorous birds such as bald eagles appear to be at high risk, primarily because of the high concentrations of PCBs in fish. Risks to great horned owl and robin are somewhat lower, however, HQs for these species are greater than 1, and thus are at moderate risk via dietary exposure pathways. However, the ERA indicated that great horned owls may be at greater risk than the dietary HQ indicates, based on measured PCB concentrations in great horned owl eggs collected in the KRE (Camp Dresser & McKee, 2003b; see Section 7.6.3 of this document for an evaluation of PCB concentrations in bird eggs). Studies on great horned owl nesting success and contaminant exposure in the Kalamazoo River area are currently being conducted by the PRPs.

Table 7.10. Summary of MDEQ ERA dietary-based HQ results for birds

Species	No effect TRV-based hazard quotient[a]	Low effect TRV-based hazard quotient[a]
Robin	2.3	1.8
Bald eagle	5.4	4.3
Great horned owl	5.0	1.7

a. Risks are summarized as hazard quotients, which are calculated by dividing the estimated daily exposure concentration by the no effect or low effect TRV. Hazard quotients greater than 1 indicate exposure greater than TRVs.

Source: Camp Dresser & McKee, 2003b.

7.5 Bald Eagle Reproductive Success

7.5.1 Data sources

- Bald eagle nesting observations made by the U.S. FWS (Dave Best, U.S. FWS, personal communication, 2000, 2001, 2002, 2003).
- Data on PCB concentrations in bald eagle eggs collected by Dr. Charles Mehne (2000), the U.S. FWS (Best, 2002), and Michigan State University Aquatic Toxicology Laboratory (2002e).
- Data on PCB concentrations in bald eagle nestling blood plasma collected in 1999 (Summer et al., 2002).

Bald eagle nesting in the Kalamazoo River area and elsewhere in Michigan has been monitored since 1960 (Camp Dresser & McKee, 2001). Bald eagle nesting pairs have been observed annually in the Allegan State Game Area since 1990 (Dave Best, U.S. FWS, personal communication, 2000, 2001). During this time, attempted nests have been inventoried and the number of young successfully reared per occupied nest has been recorded.

Four failed bald eagle eggs were collected from nests in the Allegan State Game Area in 1994 and 1996 and analyzed for total PCBs as Aroclors, the insecticide DDE, and other organochlorine pesticides at the Mississippi State Chemistry Laboratory (Mehne, 2000). An additional bald eagle egg was collected from the same area in 2000 and analyzed for total PCBs and DDE at the U.S. FWS Patuxent Analytical Control Facility (Best, 2002). An additional addled bald eagle egg was collected by Michigan State University Aquatic Toxicology Laboratory (2002e) from Ottawa Marsh in 2000 and analyzed for total PCBs as the sum of congeners.

The MDEQ collected blood plasma from two bald eagle chicks in the Highbanks Game Refuge (Allegan State Game Area) in 1999 (Summer et al., 2002). Plasma samples were analyzed for DDE and for 20 PCB congeners. These congeners include a set of 18 commonly analyzed congeners (8, 18, 28, 44, 52, 66, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, 209) as well as PCB 110 and PCB 156. The Trustees used a linear regression model to estimate total PCB concentrations from the measured sum of detected congeners.[1]

7.5.2 Benchmarks and injury thresholds

Nesting success

The mean annual bald eagle productivity rate needed to maintain a healthy population has been estimated as 1.0 young per nest (Kubiak and Best, 1991), while a rate of less than 0.7 young per nest is associated with a declining population (Sprunt et al., 1973; Meyer, 1995). Bald eagle nesting success has been observed for several locations in the Great Lakes Region. The productivity rate for all nests in inland Michigan between 1989 and 1993 was 1.0 young per nest

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1. The relationship between the sum of 20 measured congeners to total PCBs was developed from eagle and owl plasma PCB congener data from the Kalamazoo River and nearby areas in Michigan. Seventy-seven PCB congeners or co-eluting congener combinations were measured in plasma from 17 nestlings and thus a relationship could be developed from the sum of the same 20 congeners which were measured in other bald eagle plasma samples and the sum of the 77 congeners measured in plasma from these 17 nestlings. The relationship is characterized by the equation: $sum_{77\ congeners}(\mu g/kg) = 4.57 \times sum_{20\ congeners}(\mu g/kg) + 0.98$ (multiple R-squared = 0.9923, residual standard error = 16.659 μg/kg).

(Dykstra et al., 2001), equivalent to the healthy productivity rate postulated by Kubiak and Best (1991).

In the Lower Fox River/Green Bay area, elevated PCBs and DDE have caused a depression in bald eagle productivity (Dykstra et al., 2001). The mean success rate in the Fox River/Green Bay from 1987 to 1996 was 0.55 young per nest.

Egg PCB concentrations

Productivity of bald eagles is not easily evaluated in laboratory experiments, and thus it is difficult to develop unambiguous dose-response relationships for PCBs in bald eagle eggs (Elliott and Harris, 2002). Kubiak and Best (1991), Wiemeyer et al. (1993), Elliott and Harris (2002), and Nisbet and Risebrough (1994) used relationships between geospatial differences in PCB and DDE concentrations and productivity to postulate field-based toxicity thresholds for each contaminant (Table 7.11). Based largely on the Wiemeyer et al. (1993) work, egg toxicity thresholds may be > 3.0 mg/kg ww for PCBs and > 3.6 mg/kg ww for DDE. Major impacts on productivity (reductions of 50% or greater) are possible at PCB concentrations of 13-23 mg/kg ww and DDE concentrations of 3.6-6.3 mg/kg ww. While the studies listed in Table 7.11 found a statistically significant relationship between PCB concentrations in eggs and productivity, Elliott and Harris (2002) did not. Whether individual studies find statistically significant relationships or not appears to depend in part on the degree of PCB contamination being investigated (Elliott and Harris, 2002).

Table 7.11. Bald eagle egg toxicity levels identified from comparisons of regional productivities and contaminant concentrations

Productivity response	Egg PCB toxic level (mg/kg ww)	Egg DDE toxic level (mg/kg ww)	Reference
No productivity reduction		< 2.5	Nisbet and Risebrough, 1994
“Healthy” reproduction	< 1.7	< 6.0	Kubiak and Best, 1991
“Normal” productivity	< 3.0	< 3.6	Wiemeyer et al., 1993
10% productivity reduction	3.0-5.6		Wiemeyer et al., 1993
30% productivity reduction	5.6-13.0		Wiemeyer et al., 1993
30% productivity reduction	—	6 (3.6-12)	Elliott and Harris, 2002
50% productivity reduction	13-23	3.6-6.3	Wiemeyer et al., 1993
Productivity approximately halved		> 5.0	Nisbet and Risebrough, 1994
70% productivity reduction	> 23		Wiemeyer et al., 1993
75% productivity reduction		> 6.3	Wiemeyer et al., 1993

The U.S. FWS and Stratus Consulting (1999) conducted a detailed evaluation of Great Lakes bald eagle nesting success for the Lower Fox River/Green Bay NRDA. Their analysis, which was based on bald eagle nest success and PCB concentration data for nests in Michigan and Wisconsin from 1986 through 1997, concluded that the probability of bald eagle nest success measurably decreases at egg PCB concentrations greater than approximately 20 mg/kg (fresh ww) (Figure 7.4). This threshold concentration is generally consistent with the threshold ranges listed above in Table 7.11 and is reasonably consistent with studies of other species which indicate that thresholds for PCBs in eggs are higher than thresholds for DDE (Elliott and Harris, 2002). Furthermore, in their comprehensive review of the effects of PCBs on bald eagle reproduction, Elliott and Harris (2002) concurred with the 20 mg/kg (fresh ww) value derived in the Lower Fox River/Green Bay NRDA. Therefore, the Stage I Injury Assessment uses a PCB egg concentration of 20 mg/kg (fresh ww) as an injury threshold for bald eagles. This value is close to the upper end of the range (23 mg/kg) from Wiemeyer et al. (1993) for 50% reduction in egg hatching success. The DDE concentration that corresponds to this effect level is 6.3 mg/kg (fresh ww) (Wiemeyer et al., 1993).

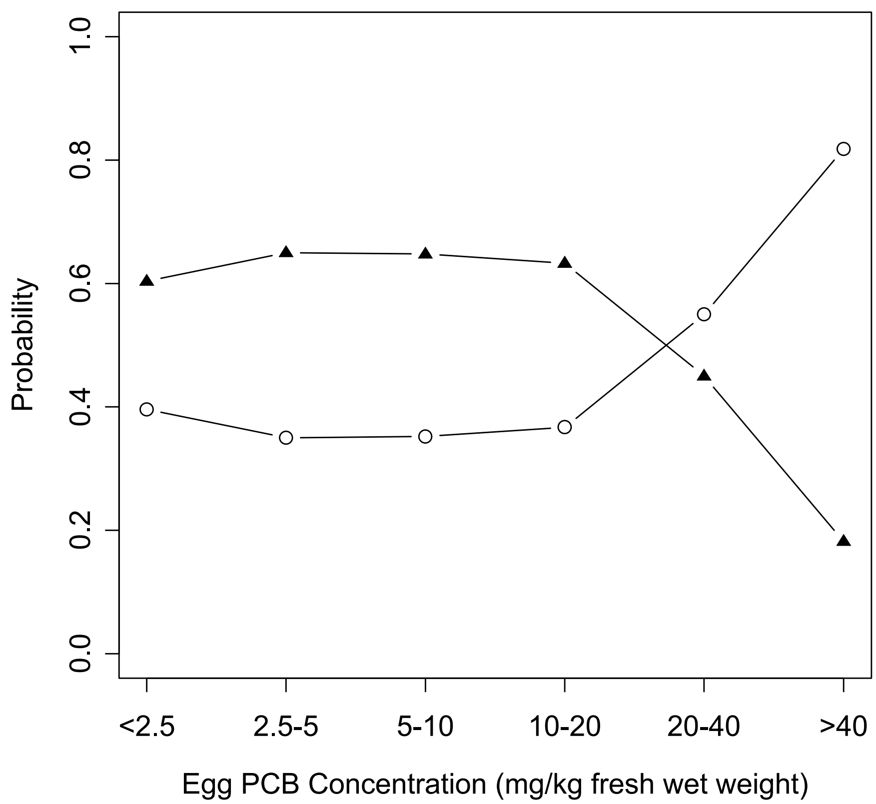


Figure 7.4. Probability of bald eagles in inland Michigan and Wisconsin and Green Bay producing no young (open circles) or one or more young (triangles) in relation to egg PCB concentrations, 1986-1997.

Source: U.S. FWS and Stratus Consulting, 1999.

Nestling plasma concentrations

There are no reliable laboratory TRVs available for the toxicological evaluation of PCB concentrations in bald eagle nestling plasma. However, research has shown that concentrations of both PCBs and DDE in nestling plasma collected in the field are associated with reduced productivity in distinct populations of bald eagles (Bowerman et al., 2003). Nestling plasma PCB concentrations can be used to compare the PCB exposure of Kalamazoo River bald eagle nestlings to the exposure of nestlings in other locations.

Bowerman et al. (2003) measured PCB and DDE concentrations in bald eagle plasma in 10 Great Lakes subpopulations. Plasma samples were collected between 1987 and 1992 and productivity was evaluated from data collected from 1977 to 1992. Geometric mean PCB concentrations in plasma of nestlings from Great Lakes subpopulations were significantly greater ($p = 0.0001$) than those from Voyageurs National Park in Minnesota, or from other interior Michigan or Minnesota subpopulations. Geometric mean DDE concentrations in plasma of nestlings from Great Lakes subpopulations and from Voyageurs National Park were significantly greater ($p = 0.0001$) than those from other interior Michigan or Minnesota subpopulations. Across all the data, productivity was inversely correlated with geometric mean PCB and DDE concentrations in plasma. Based on the regressions of productivity against plasma PCB and DDE concentrations, a productivity rate of 1.0 young per occupied nest was associated with 35 μg/kg PCB and 11 μg/kg DDE, and a productivity rate of 0.7 was associated with 125 μg/kg PCB and 28 μg/kg DDE.

Nestling plasma total PCB concentrations from monitoring programs averaged 546 μg/kg ww from nests along Lake Michigan, 42.6 μg/kg ww in inland Wisconsin, and 18.35 μg/kg ww in the inland Lower Peninsula, Michigan (Dykstra et al., 1998; Summer et al., 2002). The mean concentration of total PCBs in nestling bald eagle plasma in the Lower Fox River and Green Bay, Wisconsin areas was 267 μg/kg ww and 207 μg/kg ww, respectively (Dykstra et al., 1996, 2001). Individual nestling plasma concentrations from the Lower Fox River/Green Bay ranged from 83 to 901 μg/kg ww. The U.S. FWS concluded that bald eagles in this area are exposed to PCBs at concentrations sufficient to cause injury based on reduced productivity (U.S. FWS and Stratus Consulting, 1999).

Because of a general shift from collecting bald eagle eggs to collecting bald eagle plasma, Elliott and Harris (2002) developed a relationship between plasma concentrations of PCBs and DDE using regional mean concentrations for eggs and plasma from the Pacific Northwest and the Great Lakes. Using their relationship for PCBs, a plasma concentration of 189 μg/kg PCB is associated with the threshold of 20 mg/kg ww in eggs described in the previous section of this document. This plasma concentration is slightly higher than the 125 μg/kg PCB in plasma that Bowerman et al. (2003) demonstrated as corresponding to a productivity rate of 0.7 young per occupied nest. A plasma concentration of 44 μg/kg DDE is associated with the threshold of

6.3 mg/kg ww in eggs described in the previous section, which is also higher than the 28 μg/kg DDE threshold that Bowerman et al. (2003) shows corresponding to a productivity rate of 0.7.

7.5.3 Results

Nesting success

From 1960 to 1989, there were no known bald eagle nesting attempts along the Kalamazoo River, with the exception of a single nesting attempt in Ottawa Marsh in 1981. Nesting success for this attempt is unknown. One bald eagle pair has attempted to nest in the Ottawa Marsh since 1990, and a second pair has attempted to nest in the Highbanks Game Refuge (Highbanks Unit/Swan Creek Marsh) since 1993 (Table 7.12; see Figure 1.1). In this period of record, only six young have been successfully reared from these nests, in 1998, 1999, 2000, and 2003. Neither nest was successful in 2001. In 2002, a bald eagle pair attempted to nest in a new area near New Richmond. No eagles were produced in any of the three territories in 2002 (Dave Best, U.S. FWS, personal communication, 2002). In 2003, there were three bald eagle nesting attempts, and one young reared (Dave Best, U.S. FWS, personal communication, 2003).

Table 7.12. Bald eagle productivity in the Kalamazoo River area, 1990-2003

Year	Number of nests attempted	Number of young reared
1990	1	0
1991	1	0
1992	1	0
1993	2	0
1994	2	0
1995	2	0
1996	2	0
1997	2	0
1998	2	2
1999	2	2
2000	2	1
2001	2	0
2002	3	0
2003	3	1

Source: Dave Best, U.S. FWS, personal communication, 2000, 2001, 2002, 2003.

From 1990 through 2003, the mean productivity rate for the three Kalamazoo River bald eagle nests is only 0.2 young per nest attempt, which is a much lower rate than observed in other coastal Lake Michigan nests, inland Michigan nests, or even along the Fox River/Green Bay (Table 7.13). The Kalamazoo River bald eagle productivity rate is also one-fifth of the value generally considered to be required to maintain a healthy population, 1.0.

Table 7.13. Comparison of bald eagle nesting success in the Kalamazoo River to other locations

Location	Period of record	Average number reared per active nest
Kalamazoo River	1990-2003	0.2
Fox River/Green Bay	1974-1998	0.7
Coastal Michigan	1974-1997	0.8
Inland Michigan	1974-1997	1.0

Sources: M. Meyer, Wisconsin DNR, personal communication, March 1999; Dave Best, U.S. FWS, personal communication, 2000, 2001, 2002, 2003.

These data indicate that Kalamazoo River eagles are experiencing dramatically decreased reproductive rates compared to bald eagles in reference locations and in other locations known to be contaminated with PCBs. While bald eagle productivity has been shown to be food limited in some areas along the Lake Superior shoreline (Dykstra et al., 1998), similar studies have not found food to be a limiting factor for bald eagles nesting along the coast of Lake Michigan (Dykstra et al., 2001). Furthermore, the dramatic difference in productivity rates between the Kalamazoo River nests and other coastal Lake Michigan and inland Michigan sites indicates that factors other than food availability are most likely limiting bald eagle production in the area. As described below, exposure to PCBs is most likely at least a contributing factor to the decreased reproductive rates of Kalamazoo River bald eagles.

Egg PCB concentrations

PCB concentrations in Kalamazoo River bald eagle eggs are much higher than concentrations in eggs from other Lake Michigan coastal sites or from inland Michigan areas (Table 7.14). Compared to the average PCB concentration in eggs from other Lake Michigan coastal sites, the highest PCB concentration detected in Kalamazoo River eagle eggs, 122 mg/kg (fresh ww), is more than 7 times higher and the average PCB concentration detected in Kalamazoo River eagle eggs, 63 mg/kg (fresh ww) is approximately 4 times higher.

Table 7.14. Concentrations of total PCB and DDE in bald eagle eggs

Location	Year	Total PCB (mg/kg fresh ww)[a]	DDE (mg/kg fresh ww)[a]
Kalamazoo River (Ottawa Marsh)	1994[b]	100/122	8.2/11
	1996[b]	53/32	11/6.1
	2000	32(41)[c]	8.1
	Average	63	8.8
Other Lake Michigan coastal sites, average (range)	1994-2000	17.0 (< 0.0095-34)	4.0 (< 0.0019-8.3)
Inland Michigan lower peninsula sites, average (range)	1994-1997[d]	3.9 (0.84-7.2)	1.1 (0.42-1.75)
Stage I Injury Assessment TRV		20	6.3

a. Values represent calculated estimates of concentrations when freshly laid (Best, 2002).
b. Pairs of eggs collected in 1994 and in 1996 were collected from the same clutch, and thus are not independent samples.
c. Second value is a duplicate analysis of the same egg conducted by Michigan State University Aquatic Toxicology Laboratory (2002e).
d. No eggs were collected from inland Michigan lower peninsula sites from 1998 to 2000.
Sources: Mehne, 2000; Best, 2002; Michigan State University Aquatic Toxicology Laboratory, 2002e.

All five of the Kalamazoo River eagle eggs exceed the hatching success injury threshold of 20 mg/kg (fresh ww). The two eggs collected in 1994 are 5 and 6 times greater than the TRV, and eggs collected in other years are approximately 1.5 to 2.5 times greater. These data indicate that PCB concentrations in the Kalamazoo River bald eagle eggs are sufficient to cause the decreased egg hatching that has been observed in these nests.

Four of the five of the Kalamazoo River bald eagle eggs also have DDE concentrations that exceed the 6.3 mg/kg (fresh ww) TRV from Wiemeyer et al. (1993). However, the PCB egg concentrations are much more elevated relative to the PCB TRV than are the DDE egg concentrations relative to the DDE TRV. Although the available data are not sufficient to determine the relative contributions of PCB and DDE to the observed reductions in Kalamazoo River eagle egg hatchability, the Trustees conclude that PCB concentrations in the eggs are sufficient to cause the observed injury of reduced hatching success in the absence of DDE.

Nestling plasma PCB concentrations

Estimated total PCB concentrations in the two Kalamazoo River bald eagle nestling plasma samples collected from one nest in 1999 are similar to concentrations observed in other Lake Michigan bald eagle nestlings, and higher than those observed in inland locations (Table 7.15). The concentration of PCBs in one nestling plasma sample collected in 2000 was 773.41 µg/kg ww (Michigan State University Aquatic Toxicology Laboratory, 2002e), higher than the

Table 7.15. PCB and DDE concentrations in plasma of nestling bald eagles from the Kalamazoo River and other locations in Michigan and Wisconsin

Location	Year	Number of samples	Sum of 20 PCB congeners (µg/kg ww)	Total PCBs (µg/kg ww)	DDE (µg/kg ww)
Kalamazoo River samples					
Allegan State Game Area[a]	1999	2[b]	80.00 (63.05-96.94)	366.6 (289-444.0)	14.2 (11.9-16.5)
Allegan State Game Area[c]	2000	1	169.89	773.41	—
Mean	1999-2000	2	124.9	570.0	14.2
Geometric mean	1999-2000	2	116.6	532.5	—
Concentrations from selected locations					
Lake Michigan[a]	1999	13[d]	119.3 (11.5-297.4)	546 (53.7-1,360)	56.5 (16.0-128)
Lower Fox River, WI[e]	1991-1995	5	—	267[f] (120-547)	6[f] (< 2.5-54)
Green Bay, WI[e,g]	1987-1995	8	—	207[f] (83-901)	53[f] (4-361)
Lake Erie[h]	1987-1992	35	—	199[f] (81-1,325)	22[f] (< 5-429)
Lake Michigan[h]	1987-1992	25	—	154[f] (14-628)	35[f] (< 5-235)
Lake Superior[h]	1987-1992	45	—	127[f] (12-640)	25[f] (< 5-306)
Lake Huron[h]	1987-1992	12	—	105[f] (5-928)	25[f] (< 5-78)
Inland Wisconsin[i]	1990-1994	38	—	42.6[f] (20.1-262)	3.0[f] (< 2.5-5.5)
Eastern Upper Peninsula, Michigan[h]	1987-1992	16	—	32[f] (< 10-146)	12[f] (< 5-24)
Lower Peninsula, Michigan[h]	1987-1992	49	—	31[f] (< 10-200)	10[f] (< 5-193)
Western Upper Peninsula, Michigan[h]	1987-1992	48	—	25[f] (< 10-177)	10[f] (< 5-245)
Inland Lower Peninsula, Michigan[a]	1999	32	3.8 (ND-31.0)	18.35 (ND-143)	4.5 (ND-18.3)

Table 7.15. PCB and DDE concentrations in plasma of nestling bald eagles from the Kalamazoo River and other locations in Michigan and Wisconsin (cont.)

Location	Year	Number of samples	Sum of 20 PCB congeners (μg/kg ww)	Total PCBs (μg/kg ww)	DDE (μg/kg ww)
Threshold concentrations					
Egg threshold equivalent[j]	—	—	—	189	44
Reduced Great Lakes productivity threshold[k]	—	—	—	125	28

a. Summer et al., 2002. Total PCB concentration was estimated from 20 congener sum. See Section 7.5.1 for explanation.
b. Two 1999 samples from Allegan State Game Area were from same clutch and thus are not independent samples.
c. Michigan State University Aquatic Toxicology Laboratory, 2002e.
d. Does not include two samples from Allegan State Game Area.
e. Dykstra et al., 1996.
f. Geometric mean.
g. Dykstra et al., 2001.
h. Bowerman et al., 2003.
i. Dykstra et al., 1998.
j. Calculated from egg thresholds presented in Section 7.5.2, using relationships developed by Elliott and Harris (2002).
k. Bowerman et al., 2003. Based on relationships between productivity and geometric mean concentrations in plasma of nestling bald eagles within nine subpopulations in the upper Midwest.

concentrations observed at the Lower Fox River and Green Bay, an area also contaminated with PCBs (see Table 7.15). Kalamazoo River nestling plasma PCB concentrations are also higher than the concentration in Great Lakes bald eagle plasma of 125 μg/kg shown by Bowerman et al. (2003) to be correlated with a reproductive rate of 0.7. DDE concentrations in the two plasma samples collected in 1999 were generally lower than those observed in other Lake Michigan bald eagle populations with similar PCB concentrations. They were also lower than the estimated threshold DDE concentration of 28 μg/kg DDE developed by Bowerman et al. (2003). The Kalamazoo River eagle nestling plasma data confirm that nestlings are exposed to PCBs and that their plasma contains concentrations of PCBs that exceed injury thresholds for reduced productivity.

7.6 PCB Concentrations in Eggs of Other Birds

7.6.1 Data sources

The following data sources were used to evaluate reproductive injury to birds other than bald eagles in this section:

- Data on PCB congener concentrations in bird eggs collected by Stratus Consulting in 1995 (A.D. Little, 1996; Midwest Research Institute, 1996)
- Data on PCB concentrations in bird eggs collected by Dr. Charles Mehne (Mehne, 2000)
- Data on PCB concentrations in bird eggs compiled by the MDNR (1987b)
- Data on PCB concentrations in bird eggs collected by Michigan State University Aquatic Toxicology Laboratory (2002d; 2002e).

Stratus Consulting staff sampled bird eggs from five locations in the KRE in 1995 as part of a Trustee study. Samples included one great horned owl egg, five red-winged blackbird eggs, two robin eggs, one wood duck egg, one wood thrush egg, and one yellow warbler egg. These egg samples were analyzed for a suite of 45 PCB congeners (analytical data in A.D. Little, 1996). Split samples from five of these eggs were also analyzed separately for coplanar PCB congeners (PCB 77, 81, 126, and 169) and dioxins/furans (analytical data in Midwest Research Institute, 1996). These results were compiled by Stratus Consulting using the results from Midwest Research Institute for coplanar PCB congeners where available. The sum of the 45 congeners analyzed was assumed to represent the total PCB concentration in these samples.

Additional bird egg PCB and DDE data were obtained from Mehne (2000), the MDNR (1987b), and Michigan State University Aquatic Toxicology Laboratory (2002d, 2002e). Sixteen egg samples from great blue herons, red tailed hawks, great horned owls, and wood ducks were collected from locations in the Allegan State Game Area in 1993 and 1994 and analyzed for total PCBs as Aroclors at the Animal Health Diagnostic Laboratory in Lansing, Michigan (1993 samples) or at the Illinois Department of Agriculture in Centralia, Illinois (1994 samples; Mehne, 2000). Additionally, 14 mute swan eggs were collected in 1986 from the Allegan State Game Area and analyzed for total PCBs (MDNR, 1987b). Two great horned owl eggs were collected in 2002 from the Allegan State Game Area and 3 eastern bluebird eggs, 11 house wren eggs, and 6 tree swallow eggs were collected from Trowbridge in 2001 and analyzed for total PCBs as the sum of congeners (Michigan State University Aquatic Toxicology Laboratory, 2002d, 2002e).

7.6.2 Toxicological benchmarks

Total PCBs

The adverse effects of PCBs in eggs have been determined in both laboratory exposure experiments and field assessments that link measurement endpoints with egg PCB concentrations. Chickens, a particularly sensitive species, experience reduced hatching success, embryo or chick deformities, and other reproductive impairment at concentrations of 1.5 to 5 mg PCB/kg egg ww. In other species of birds, LOELs for the most sensitive endpoint measured range upward from approximately 4 mg PCB/kg egg ww (Table 7.16). NOELs range upward from 1.3 mg PCB/kg egg ww. The data in this table demonstrate the large differences between species' sensitivities to PCBs. For example, the mallard LOEL reported in Table 7.16 exceeds that of the chicken by a factor of more than 50. Many of the study results listed in Table 7.16 may be confounded by the fact that they are based on field studies in which parameters other than PCBs (e.g., other contaminants, hatching and rearing conditions) could not be controlled. Hoffman et al. (1995) selected a TRV range of 8 to 25 mg/kg ww total PCBs in eggs of terns, cormorants, and eagles for the endpoint of reduced hatching success. This TRV range is used for this injury evaluation. Sensitive species may experience reproductive effects at concentrations from 1.5 to 5 mg PCB/kg egg ww.

Table 7.16. Egg total PCB concentrations causing adverse effects

Species	PCB	LOEL mg/kg ww	NOEL mg/kg ww	Adverse effect[a]	Laboratory (L) or field (F) study	Reference
Chicken	Total PCB	1.5	0.95	H	L	Britton and Huston, 1973
Chicken	Total PCB	2.5	0.36	H	L	Scott, 1977
Chicken	Total PCB	4	—	D, H	L	Tumasonis et al., 1973
Bald eagle	Total PCB	4	—	S	F	Ludwig et al., 1993
Caspian tern	Total PCB	4.2	—	S	F	Yamashita, 1993
Chicken	Total PCB	5	< 5	P, F	L	Platonow and Reinhart, 1973
Chicken	A1242	6.7[b]	0.67[b]	G	L	Gould et al., 1997
Chicken	A1254	6.7[b]	0.67[b]	G	L	Gould et al., 1997
Common tern	Total PCB	7	5.2-5.6	H	F	Becker et al., 1993
Bald eagle	Total PCB	7.2	1.3	S	F	Wiemeyer et al., 1984

Table 7.16. Egg total PCB concentrations causing adverse effects (cont.)

Species	PCB	LOEL mg/kg ww	NOEL mg/kg ww	Adverse effect[a]	Laboratory (L) or field (F) study	Reference
Common tern	Total PCB	8	7	S	F	Bosveld and Van den Berg, 1994
Common tern	Total PCB	10	4.8[c]	D, H	L	Hoffman et al., 1993
Bald eagle	Total PCB	13	—	S	F	Wiemeyer et al., 1993
Ringed turtle dove	A1254	16	—	H	L	Peakall and Peakall, 1973
Forster's tern	Total PCB	19	7[c]	S	F	Bosveld and Van den Berg, 1994
Bald eagle	Total PCB	~20	—	S	F	U.S. FWS and Stratus Consulting, 1999
Forster's tern	Total PCB	22.2	4.5	H	F/L	Kubiak et al., 1989
American kestrel	Mixture of Aroclors	34.0	—	P	L	Fernie et al., 2001
Mallard	A1242	105	—	T	F	Haseltine and Prouty, 1980

a. D = embryo or chick deformities; F = reduced fertility; G = reduced chick growth; H = reduced hatching success; P = reduced egg production; S = reduced overall reproductive success; T = egg shell thinning.
b. Concentration is reported as mg/kg, but may actually be in units of μg/kg (K. Grasman, Wright State University, personal communication, 2001).
c. Based on no apparent adverse effects in field population.

TCDD-equivalents

To account for the variations in toxicity of different PCB congeners, potency can be expressed relative to the potency of TCDD, the most toxic, halogenated aromatic compound, using a TEF (see Section 6.3.2 of this document).

As was done for fish, an international group of toxicology experts developed bird TEFs for several PCB and dioxin/furan congeners (Table 7.17). These values were derived from multiple studies and are not specific to any single bird species. The TEFs can be used to calculate TCDD-eq in a sample by multiplying the concentration of each PCB congener by its TEF, and summing the results across all the congeners.

Table 7.17. Avian toxic equivalency factors of PCB, dioxin, and furan congeners relative to TCDD

Compound	Avian TEF
PCB congener 77	0.05
PCB congener 81	0.1
PCB congener 105	0.0001
PCB congener 114	0.0001
PCB congener 118	0.00001
PCB congener 123	0.00001
PCB congener 126	0.1
PCB congener 156	0.0001
PCB congener 157	0.0001
PCB congener 167	0.00001
PCB congener 169	0.001
PCB congener 189	0.00001
2,3,7,8 TCDD	1
2,3,7,8 TCDF	1
1,2,3,7,8 PeCDD	1
1,2,3,7,8 PeCDF	0.1
2,3,4,7,8 PeCDF	1
1,2,3,4,7,8 HxCDD	0.05
1,2,3,6,7,8 HxCDD	0.01
1,2,3,7,8,9 HxCDD	0.1
1,2,3,4,7,8 HxCDF	0.1
1,2,3,6,7,8 HxCDF	0.1
1,2,3,7,8,9 HxCDF	0.1
2,3,4,6,7,8 HxCDF	0.1
1,2,3,4,6,7,8 HpCDD	0
1,2,3,4,6,7,8 HpCDF	0.01
1,2,3,4,7,8,9 HpCDF	0.01
OCDD	0.0001
OCDF	0.0001

Source: Van den Berg et al., 1998.

Numerous studies have been conducted on adverse effects caused by PCB congeners and TCDD in bird eggs. Using TEFs, the PCB congener concentrations reported in these studies have been converted to TCDD-eq in Table 7.18. As the data in Table 7.18 show, chickens are particularly sensitive to TCDD toxicity compared to other bird species studied to date. Egg LD50 concentrations for chickens range from 0.04 to 0.43 µg/kg TCDD-eq, whereas LD50 concentrations for other species range from 1.4 to > 250 µg/kg TCDD-eq (laboratory studies only).

Excluding chicken, reported egg LOEL values from laboratory studies range from 0.23 µg/kg TCDD-eq (for edema in American kestrels) to 4.4 µg/kg TCDD-eq (for embryomortality and deformities in common terns). Therefore, the Trustees select a concentration range of 0.2 to 4 µg/kg TCDD-eq as an estimated range of injury threshold concentrations in bird eggs.

For chickens, the reported egg LOEL values from laboratory studies range from 0.006 µg/kg TCDD-eq to 0.32 µg/kg TCDD-eq. The Trustees select a lower injury threshold range of 0.01 to 0.04 µg/kg TCDD-eq in the egg from the lower end of the range of these LOEL values based on the assumption that differences in LOEL values across the different chicken studies listed in the table are primarily a result of the dosing concentrations and study designs used in the different studies, not because of inherently different sensitivities to TCDD toxicity among the chickens tested. It is the lower end of the reported LOEL range that thus is most relevant for selecting an injury threshold for sensitive species. The value of 0.04 is used as the upper end of the injury threshold range because it corresponds to the lowest reported LD50 value for chicken.

7.6.3 Results

Total PCBs

Measured PCB concentrations in KRE eggs of several avian species were within or above the TRV range for embryomortality (Figure 7.5). Measured concentrations ranged from 8.4 to 14.5 mg/kg ww in eastern bluebird eggs from Trowbridge, 1.5 to 44 mg/kg ww in blue heron eggs from the Allegan State Game Area, from 16 to 91 mg/kg ww in great horned owl eggs, from 2.0 to 8.3 mg/kg ww in house wren eggs from Trowbridge, and from 2.3 to 27 mg/kg ww in red tailed hawk eggs. Eggs from all of the species sampled, except for wood duck and yellow warbler, were within or above the TRV range for sensitive species.

2. No such species have been reported in the literature, although the number of bird species tested to date is limited.

Table 7.18. Bird egg TCDD-eq concentrations causing adverse effects

Species	Toxicant	Measurement	Egg toxicant concentration (µg/kg egg, ww)	TCDD-eq concentration (µg/kg egg, ww)[a]	Adverse effect[b]	Laboratory (L) or field (F) study	Reference
Chicken	TCDD	NOEL	0.1	0.1	H	L	Janz and Bellward, 1996
		NOEL	0.2	0.2	I	L	Peden-Adams et al., 1998
		LOEL[c]	0.006	0.006	D	L	Cheung et al., 1981
		LOEL	0.01	0.01	H	L	Verrett, 1970 (in Hoffman et al., 1996a)
		LOEL	0.04	0.04	H	L	Verrett, 1976 (in Hoffman et al., 1996a)
		LOEL	0.08	0.08	D	L	Walker and Catron, 2000
		LOEL	0.1 (yolk injected)	0.1	G	L	Henshel et al., 1997
		LOEL	0.3 (air cell injected)	0.3	G	L	Henshel et al., 1997
		LOEL	0.32 (yolk injected)	0.32	D	L	Walker et al., 1997
		LD50	0.12 (yolk injected)	0.12	H	L	Henshel et al., 1997
		LD50	0.15	0.15	H	L	Verrett, 1976 (in Hoffman et al., 1996a)
		LD50	0.15	0.15	H	L	Powell et al., 1996
		LD50	0.18 (air cell injected)	0.18	H	L	Henshel, 1993 (in Hoffman et al., 1996a)
		LD50	0.24 (air cell injected)	0.24	H	L	Allred and Strange, 1977 (in Hoffman et al., 1996a)
		LD50	0.3 (air cell injected)	0.3	H	L	Henshel et al., 1997

Table 7.18. Bird egg TCDD-eq concentrations causing adverse effects (cont.)

Species	Toxicant	Measurement	Egg toxicant concentration (µg/kg egg, ww)	TCDD-eq concentration (µg/kg egg, ww)[a]	Adverse effect[b]	Laboratory (L) or field (F) study	Reference
	PCB 77	LD50	2.6	0.13	H	L	Hoffman et al., 1998
		LD50	8.6	0.43	H	L	Brunström and Andersson, 1988
	PCB 105	LD50	2,200	0.22	H	L	Brunström, 1990
	PCB 118	LD50	8,000	0.08	H	L	Brunström, 1989
	PCB 126	LOEL	0.57	0.057	D	L	Walker and Catron, 2000
		LOEL	0.3	0.03	D	L	Hoffman et al., 1998
		LD50	0.4	0.04	H	L	Hoffman et al., 1998
		LD50	1.01	0.101	H	L	Fox and Grasman, 1999
		LD50	2.3	0.23	H	L	Powell et al., 1996
		LD50	3.2	0.32	H	L	Brunström and Andersson, 1988
	PCB 156	LD50	1,500	0.15	H	L	Brunström, 1990
	PCB 157	LD50	2,500	0.25	H	L	Brunström, 1990
	PCB 167	LD50	> 4,000	> 0.04	H	L	Brunström, 1990
	PCB 169	LD50	170	0.17	H	L	Brunström and Andersson, 1988
Osprey	TCDD-eq	NOEL	0.14	0.14	S	F	Woodford et al., 1998
	TCDD-eq	NOEL	~0.05	~ 0.05	H	F	Elliott et al., 2001
Bald eagle	TCDD-eq	NOEL	0.2	0.2	S	F	Elliott et al., 1996
Bobwhite	PCB 126	LD50	24	2.4	H	L	Hoffman et al., 1995

Table 7.18. Bird egg TCDD-eq concentrations causing adverse effects (cont.)

Species	Toxicant	Measurement	Egg toxicant concentration (µg/kg egg, ww)	TCDD-eq concentration (µg/kg egg, ww)[a]	Adverse effect[b]	Laboratory (L) or field (F) study	Reference
Caspian tern	TCDD-eq	NOEL	0.75	0.75	H	F	Ludwig et al., 1993
Domestic pigeon	TCDD	LOEL	3	3	G, H	L	Janz and Bellward, 1996
Eastern bluebird	TCDD	NOEL	1	1	B	F	Thiel et al., 1988
		LOEL	10	10	B	F	
Common tern	PCB 126	LOEL	44	4.4	D, H	L	Hoffman et al., 1998
		LD50	104	10.4	H	L	
	TCDD-eq	NOEL	< 4	< 1 (assuming 25% lipid)	H	L	Bosveld and Van den Berg, 1994
Double-crested cormorant	PCB 126	LD50	158	16	H	L	Powell et al., 1997
		LD50	177	18	H	L	Powell et al., 1998
	TCDD	LOEL	4	4	H	L	Powell et al., 1997
		LD50	4	4	H	L	Powell et al., 1998
	TCDD-eq	LD50	~0.55	0.55	H	F	Tillitt et al., 1992
Forster's tern	TCDD-eq	NOEL	0.05[d]	0.05	H	L/F	Kubiak et al., 1989
		LOEL	0.55[d]	0.55	H	L/F	
Great blue heron	TCDD	NOEL	2	2	H	F	Janz and Bellward, 1996
	TCDD-eq	NOEL	0.02	0.02	G, D	F	Hart et al., 1991
		LOEL	0.245	0.245			
Ring-necked pheasant	TCDD	LOEL	1 (yolk sac injected)	1	H	L	Nosek et al., 1993
		LOEL	1 (albumen injected)	1	H	L	
		LD50	1.4 (albumen injected)	1.4	H	L	
		LD50	2.2 (yolk sac injected)	2.2	H	L	
	PCB 77	NOEL	100	5	H	L	Brunström and Reutergårdh, 1986

Table 7.18. Bird egg TCDD-eq concentrations causing adverse effects (cont.)

Species	Toxicant	Measurement	Egg toxicant concentration (µg/kg egg, ww)	TCDD-eq concentration (µg/kg egg, ww)[a]	Adverse effect[b]	Laboratory (L) or field (F) study	Reference
Wood duck	TCDD-eq	NOEL	# 5	# 5	H	F	White and Seginak, 1994; White and Hoffman, 1995
		LOEL	> 20-50	> 20-50	H	F	
American kestrel	PCB 77	LD50	316	15.8	H	L	Hoffman et al., 1998
	PCB 126	LOEL	2.3	0.23	D	L	
		LD50	65	6.5	H	L	
Turkey	PCB 77	LD50	~800	40	H	L	Brunström and Lund, 1988
Black-headed gull	PCB 77	LD50	< 1,000	< 50	H	L	Brunström and Lund, 1988
Herring gull	PCB 77	LD50	> 1,000	> 50	H	L	Brunström, 1988
	TCDD-eq	NOEL	1-2	1-2	H	F	Ludwig et al., 1993
Domestic goose	PCB 77	LD50	> 1,000	> 50	H	L	Brunström, 1988
Goldeneye	PCB 77	LD50	> 1,000	> 50	H	L	Brunström and Reutergårdh, 1986
Mallard	PCB 77	LD50	> 5,000	> 250	H	L	Brunström, 1988

a. Calculated using TEFs (Van den Berg et al., 1998).
b. A = adult mortality; B = reproductive behavior; D = deformities; F = female fertility; G = chick growth; H = hatching success; I = immunological changes; M = male fertility; P = egg production; S = population size or reproductive success; T = egg shell thinning. Data are organized by the general rank order of TCDD-eq toxicity values.
c. This LOEL is an effective dose associated with a 20% increase in heart deformities (ED-20) derived from a statistically significant log dose response regression.
d. TCDD-eqs from Kubiak et al., 1989, were originally reported using TEFs from Sawyer and Safe (1982) based on testing in rats as NOEL = 0.2 and LOEL = 2.2. TCDD-eqs were recalculated using the WHO avian TEFs (Table 7.17).

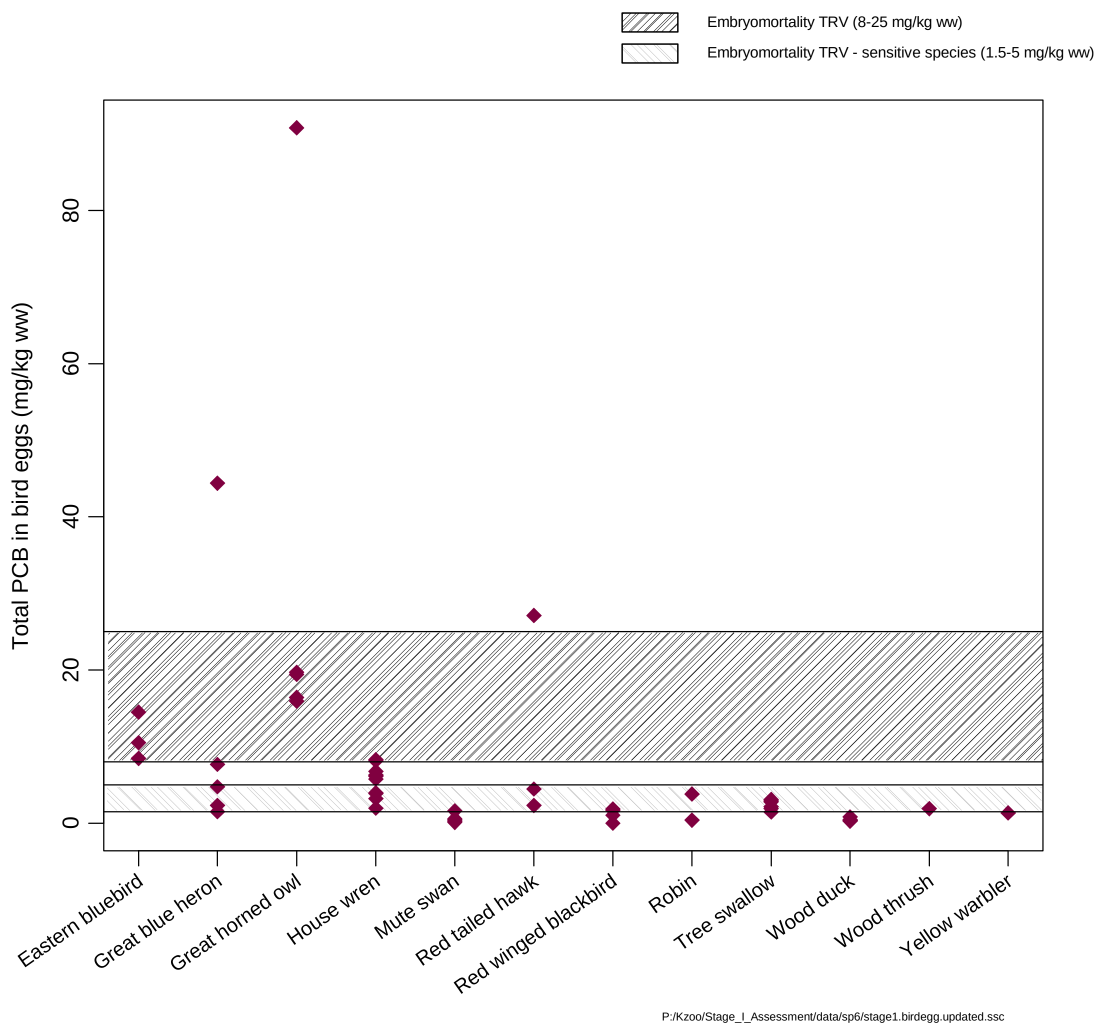


Figure 7.5. Total PCB concentrations in bird eggs collected in Allegan State Game Area and other locations along the Kalamazoo River.

Sources: Michigan Department of Natural Resources, 1987b; A.D. Little, 1996; Midwest Research Institute, 1996; Mehne, 2000; Michigan State University Aquatic Toxicology Laboratory, 2002c.

TCDD-equivalents

Measured TCDD-eq concentrations from PCBs in KRE bird eggs were above the range of embryomortality for sensitive species in 5 of 11 samples (3 red-winged blackbirds, 1 wood duck, and 1 great horned owl), and within the range in 1 red-winged blackbird sample (Figure 7.6). The maximum TCDD-eq from PCBs was 0.175 μg/kg in 1 great horned owl egg collected near Lake Allegan Dam. TCDD-eqs in other red-winged blackbird, robin, wood thrush, and yellow warbler eggs were less than 0.0074 μg/kg. TCDD-eqs from dioxins and furans were analyzed for four of the eggs sampled. PCBs accounted for 66 to 95% of the total TCDD-eq in these eggs because of the much higher concentrations of PCBs than dioxins or furans in these samples.

These results indicate that at least some avian species in addition to bald eagles (discussed in Section 7.5 of this document) may be injured by PCBs. Concentrations of total PCBs and TCDD-eqs in the eggs of several species are within the range associated with embryomortality in sensitive species. Some great blue heron, great horned owl, and red tailed hawk eggs also have PCB concentrations within the range associated with embryomortality in less sensitive species. However, the limited nature of these data make it difficult to identify the temporal and geographical extent of injury.

7.7 Total PCBs in Mammalian Diets

In this section, the results of the MDEQ ERA for mammals exposed to PCBs in their diet are discussed, and PCB concentrations in KRE fish are compared to TRVs for piscivorous mammals such as mink.

7.7.1 Data sources

Some of the same data sources used to evaluate injury to birds in Section 7.4 are also used to evaluate injury to mink in this section. As for avian species (see Section 7.4.1), the MDEQ calculated estimated exposure concentrations in the diets of mammalian species of concern based on dietary exposure models (Camp Dresser & McKee, 2003b). Mammalian species evaluated in the ERA were mink, white footed/deer mouse, muskrat, and red fox. Using dietary no effect and low effect values derived from the literature, MDEQ then calculated no effect – and low effect hazard quotients to characterize risk from PCB exposure for these species. PCB concentrations measured in KRE whole-body fish are compared to the dietary TRVs for mink. These data are described in Section 7.4.1.

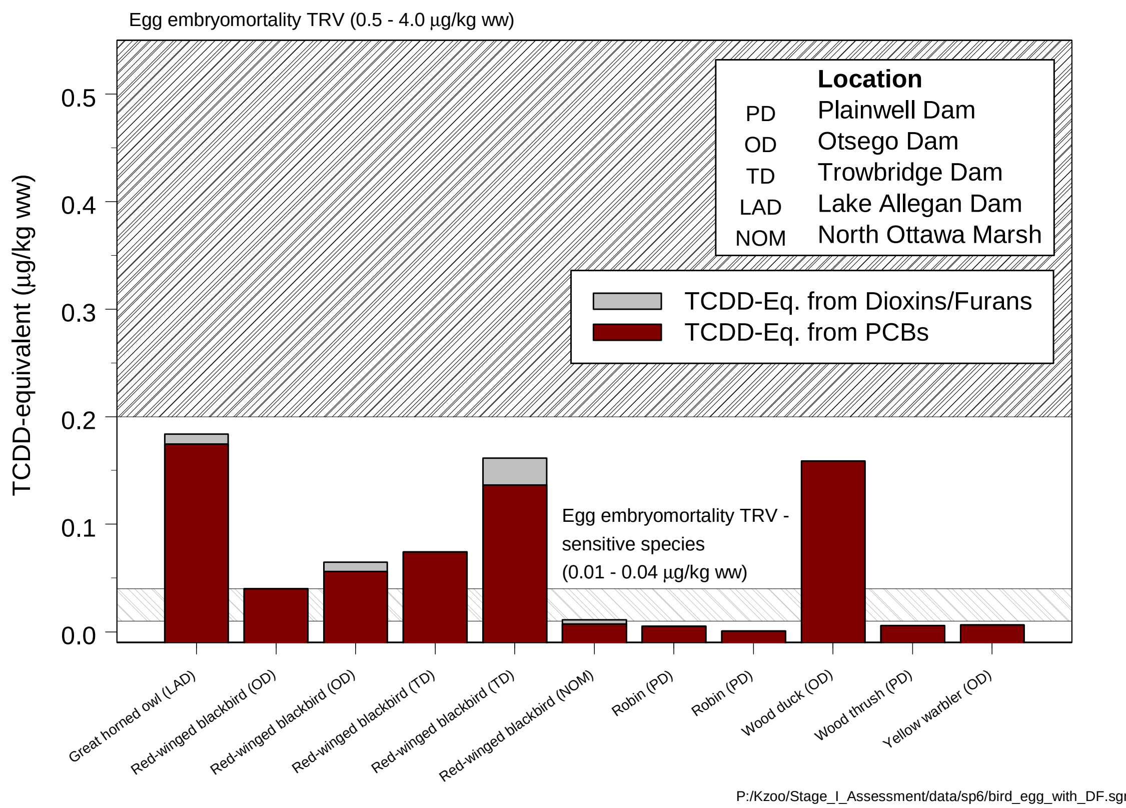


Figure 7.6. TCDD-eq concentrations based on PCBs and dioxins/furans measured in individual bird eggs in the KRE compared to TRVs for egg embryomortality. Dioxins and Furans were only measured in four samples (one great horned owl and three red winged blackbird). Contributions to toxicity by dioxins and furans in the remaining samples is unknown.

Sources: A.D. Little, 1996; Midwest Research Institute, 1996.

7.7.2 Toxicological benchmarks

Stage I Injury Assessment TRVs were developed for mink only, since mink are expected to be highly exposed to PCBs through their diet of fish and other aquatic biota and they are sensitive to PCB toxicity. The mink TRVs were developed from controlled laboratory dietary toxicity tests reported in the literature where PCBs were administered in mink diets. In many of these studies, measurement endpoints included sublethal effects such as increased organ weights, altered enzyme activity, and depressed growth. Reproductive effects such as reduced or eliminated reproduction, reduced kit weight and growth, and kit mortality were also monitored.

In adult mink, female reproductive endpoints are generally the most sensitive endpoints to PCB toxicity (Peterson et al., 1993). Dietary PCB concentrations as low as 0.25 mg/kg diet ww have been found to cause delayed estrus and a reduced whelping rate in mink fed PCB contaminated carp from the Great Lakes (Restum et al., 1998). Hornshaw et al. (1983) and Platonow and Karstad (1973) found more severe reproductive effects such as reduced reproduction and kit mortality at LOELs of 0.66 and 0.64 mg/kg diet ww, respectively. The Hornshaw et al. (1983) study fed adult females PCB contaminated fish from the Great Lakes, while the Platonow and Karstad (1973) study fed mink beef from cattle that had been fed a diet laden with PCBs. Similar effects on kit survival were seen by Heaton et al. (1995a) at a LOEL of 0.72 mg/kg diet ww, and by Wren et al. (1987a, 1987b), Aulerich and Ringer (1977), and Restum et al. (1998) at a LOEL of 1.0 mg/kg diet ww. Complete elimination of reproduction is associated with dietary PCB concentrations of 1.5 mg/kg diet ww (Hornshaw et al., 1983). The low effect value for mink used in the Ecological Risk Assessment (Camp Dresser & McKee, 2003b) was 1.1 mg/kg ww. For this Stage I Assessment, a mink dietary TRV range of 0.5 to 1.0 mg/kg ww total PCBs in diet is used to evaluate injury to mink through dietary exposure. Concentrations in diet within this range would be expected to cause reduced reproduction or kit mortality in mink. Although TCDD-eq based dietary TRVs for mink are also available (e.g., Brunström et al., 2001), almost all of the available KRE PCB concentration data for potential mink dietary items, such as fish, are quantified as total PCBs or Aroclor mixtures, making TCDD-eq based TRVs difficult to use.

7.7.3 Results

The MDEQ (Camp Dresser & McKee, 2003b) calculated risks to mammals through dietary exposure by dividing the average estimated daily exposure concentration by NOEL and LOEL values (Table 7.19). The ERA concluded that risks to sensitive piscivorous predators, such as mink, are high. MDEQ concluded that carnivorous mammals such as red fox are not likely to be at significant risk unless foraging is concentrated in riparian areas with contaminated floodplain sediment and the fox diet consists of prey that have taken up substantial amounts of PCBs. Omnivorous species such as mice are not likely to be at risk through dietary exposure pathways

Table 7.19. Summary of risks to terrestrial wildlife from the ERA (MDEQ, 2003)

Species	NOEL-based hazard quotient[a]	LOEL-based hazard quotient[a]
Mink	19	15
White footed/deer mouse	0.7	0.2
Muskrat	0.3	0.08
Red fox	2.5	0.5

a. Risks are summarized as hazard quotients, which are calculated by dividing a NOEL or LOEL by the estimated daily exposure concentration. The mink NOEL is the exposure concentration predicted to cause a 10% decrease in reproduction, and the mink LOEL a 25% decrease (Camp Dresser & McKee, 2003b). For the other species, the effect endpoints vary. NOEL or LOEL values greater than 1 mean that dietary PCB exposure exceeds the TRV.

Source: Camp Dresser & McKee, 2003b.

unless they reside in the most contaminated areas. The hazard quotients for muskrat, a semi-aquatic herbivorous mammal, are also low. However the TRVs for muskrat were based on threshold values for rats which may not be equally sensitive to PCBs via ingestion. Overall, the MDEQ ERA concludes that the KRE has been, and is currently being adversely affected by PCBs.

A comparison of whole fish PCB concentrations with TRVs for mink dietary exposure is consistent with the conclusions of the ERA (Figure 7.7). Many of the samples from all years are within or above the range expected to cause reproductive toxicity to mink. Overall, 199 of the 228 fish samples (87%) collected are within or above the mink TRV range. Fish containing PCBs sufficient to cause reproductive effects in mink are distributed throughout the KRE, including Portage Creek and the Kalamazoo River from the city of Kalamazoo to Lake Michigan. These conditions have been observed since as early as 1993 and have persisted through 2000. However, it is likely that fish have accumulated PCBs since the time of the initial releases, and also likely that concentrations were much higher in the past than those measured in 1993.

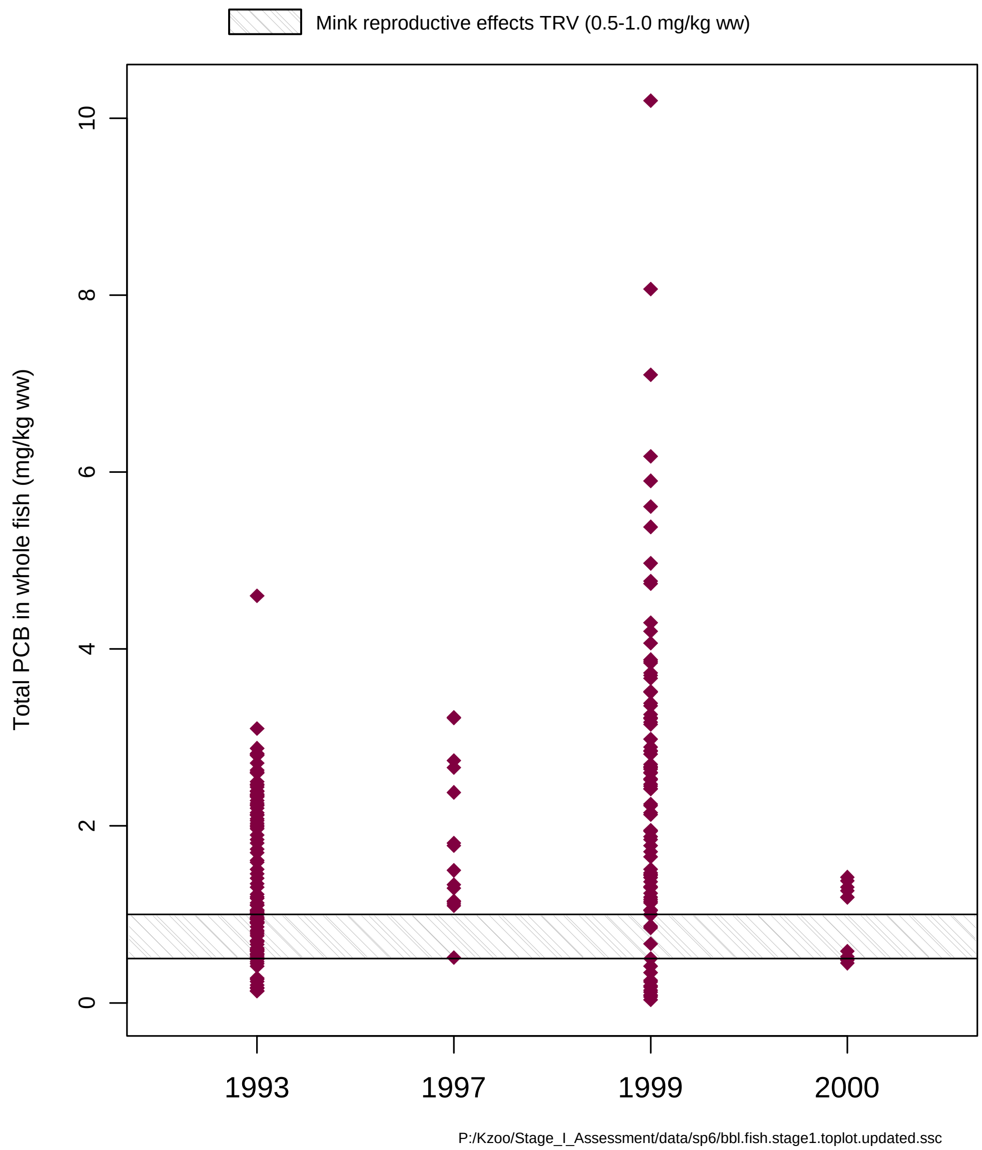


Figure 7.7. Total PCBs in whole body fish samples collected in the Kalamazoo River downstream of PRPs compared to TRVs for reproductive effects in mink. Species include channel catfish, golden redhorse, northern hogsucker, smallmouth bass, spotted sucker, and white sucker.

Sources: Blasland, Bouck & Lee 2001; Camp Dresser & McKee, 2002b; Michigan State University Aquatic Toxicology Laboratory, 2002j.

7.8 Mink Trapping Success and Total PCBs in Mink Tissue

7.8.1 Data sources

The following data sources are used to evaluate injury to mink in this section:

- Data on mink trapping success and PCB concentrations in mink liver and carcass samples collected in 1993 and 1994 by Camp Dresser & McKee (1997)
- Data on PCB concentrations in mink liver samples collected by Michigan State University Aquatic Toxicology Laboratory (2002b, 2002c).

Camp Dresser & McKee (1997) collected mink from one reference location (Battle Creek) and four assessment locations (near Plainwell Dam, Otsego Dam, Trowbridge Dam, and Lake Allegan Dam) in riparian areas of the Kalamazoo River between December 1993 and April 1994. Trapping success at each location was recorded and can be expressed as number of mink per "trap night." The number of nights that the traps were set varied between locations, and many of these nights were not considered to be "effective trap nights" because of high water levels or other logistical problems. Therefore, the number of effective trap nights was 10 or 11 at each sampling area. Liver and carcass samples were collected from the mink and analyzed for total PCBs as Aroclors. Concentrations on a wet weight basis were calculated using the reported dry weight basis concentrations and moisture content in samples.

Additional mink liver data were collected by Michigan State University Aquatic Toxicology Laboratory (2002b, 2002c) between 1999 and 2002. Seven mink were sampled from locations in the city of Kalamazoo and near Plainwell and Otsego dams (Giesy Ecotoxicology, 2001). Another three mink were collected from D. Avenue in Kalamazoo and Trowbridge in 2001 and 2002. Three mink were captured near Fort Custer State Park, an upstream reference location in 2001 and 2002. Livers from these mink were analyzed for total PCBs as the sum of congeners.

7.8.2 Toxicity reference value derivation

Few studies have evaluated toxic effects associated with PCB concentrations in mink carcasses. Leonards et al. (1995) calculated whole body concentrations of total PCBs in mink from experimental literature data. Whole body EC_{50} concentrations (concentrations at which 50% of the subjects experience the tested effect) for reduced litter size and kit survival were estimated from these studies. The calculated EC_{50} concentration for reduced litter size was 1.2 μg/g PCB ww, in mink whole body samples and concentrations greater than 20 μg/g were associated with complete reproductive failure. The mean lipid weight of muscle was 2 to 3%, which translates the EC_{50} to 40 to 60 μg/g on a lipid normalized basis, similar to the EC_{50} of 65 μg/g (lipid) for reduced litter size reported by Kihlström et al. (1992). Leonards et al. (1995) also calculated an EC_{50} of 2.36 μg/g ww for kit survival based on predicted whole body concentrations from the results of experimental literature. The EC_{50} whole body PCB concentration of 1.2 μg/g ww for reduced litter size and the EC_{50} of 2.36 μg/g ww for kit survival are used to evaluate injury in whole body mink in this Stage I Assessment, although it should be recognized that the injuries at these concentrations are severe (50% reduction in litter size or kit survival), and thus these values are not equivalent to LOEL values.

Tissue PCB concentrations associated with adverse effects are also available for mink livers (Kannan et al., 2000). Adverse reproductive effects have been observed at adult mink liver PCB concentrations of 0.2 to 7.25 mg/kg ww (Platonow and Karstad, 1973; Den Boer, 1984; Wren et al., 1987a, 1987b; Heaton et al., 1995a; Restum et al., 1998; Halbrook et al., 1999). With the exception of one study (Halbrook et al., 1999), all the LOEL concentrations in mink livers for reproductive effects are less than 2.2 mg/kg ww. Halbrook et al. (1999) evaluated the effects of a diet of PCB contaminated fish from near Oak Ridge, Tennessee, on ranch mink, and found a LOEL for reproductive effects of 7.25 mg/kg PCBs ww in adult livers. This higher LOEL value may be the result of a different PCB congener profile or study design. A reproductive effects TRV range of 0.2 to 2.2 mg/kg ww for reproductive effects is used to evaluate injury in mink livers in this Stage I Assessment. Studies on mortality effects associated with PCBs in mink livers report LOEL concentrations from 4.2 to 12.0 mg/kg ww (Aulerich et al., 1973; Platonow and Karstad, 1973; Kubiak and Best, 1991). This range is also used to evaluate injury to mink in this Stage I Assessment.

7.8.3 Results

Trapping success

Mink trapping success was lower in assessment areas than in the upstream reference location in Battle Creek (Table 7.20). Under a similar trapping effort, five mink were caught at Battle Creek whereas no mink were captured near Otsego Dam, one was captured upstream of Plainwell Dam, and two were captured at both Trowbridge Dam and Lake Allegan Dam. Camp Dresser & McKee (1997) attributed the reduced trapping success to reduced mink populations in the sampling locations within the assessment areas. There was suitable mink habitat throughout the study area, apparently adequate food supply, and equal trapping effort among sites. This study was not designed to sample mink populations, rather to collect mink for tissue sampling; however, it provides supporting evidence that KRE mink populations are adversely affected by PCB contamination in the Kalamazoo River.

Table 7.20. Mink trapping success, 1993-1994

Sampling location	Number of mink caught	Number of effective trap nights
Battle Creek (reference)	5	10
Upstream of Plainwell Dam	1	10
Downstream of Otsego Dam	0	10
Upstream of Trowbridge Dam	2	11
Downstream of Lake Allegan Dam	2	11
Source: Camp Dresser & McKee, 1997.		

Total PCBs

PCB concentrations in mink tissue indicate that injury may be occurring; however, few samples have been analyzed. The concentrations of PCBs in all five carcasses from assessment areas are higher than the concentration associated with 50% reduction in kit survival, and four of five are higher than the concentration associated with 50% reduction in litter size (Figure 7.8). In contrast, three of the four samples from the reference location are below the concentration associated with reduced kit survival and all are lower than the concentration associated with reduced litter size.

Livers from the same mink collected in 1994 and others sampled between 1999 and 2002 show similar results (Figure 7.9). All of the livers from mink sampled in 1994 have concentrations above the range associated with reproductive effects, and the concentration of PCBs in one sample, at 12.5 mg/kg ww, is higher than the mortality effects TRV. Livers from mink collected from the KRE assessment area between 1999 and 2002 range in PCB concentration from 0.03 to 6.03 mg/kg ww, with eight out of the ten samples falling within the TRV range. The samples collected in the Fort Custer reference location have concentrations from 1.55 to 3.68 mg/kg ww.

These tissue concentration results suggest that KRE mink are injured by exposure to PCBs. This conclusion is consistent with the analysis of PCBs in KRE mink diet (Section 7.7) and the observed reduced mink trapping success. However, the tissue data are insufficient to draw any conclusions about the geographic or temporal extent of injury.

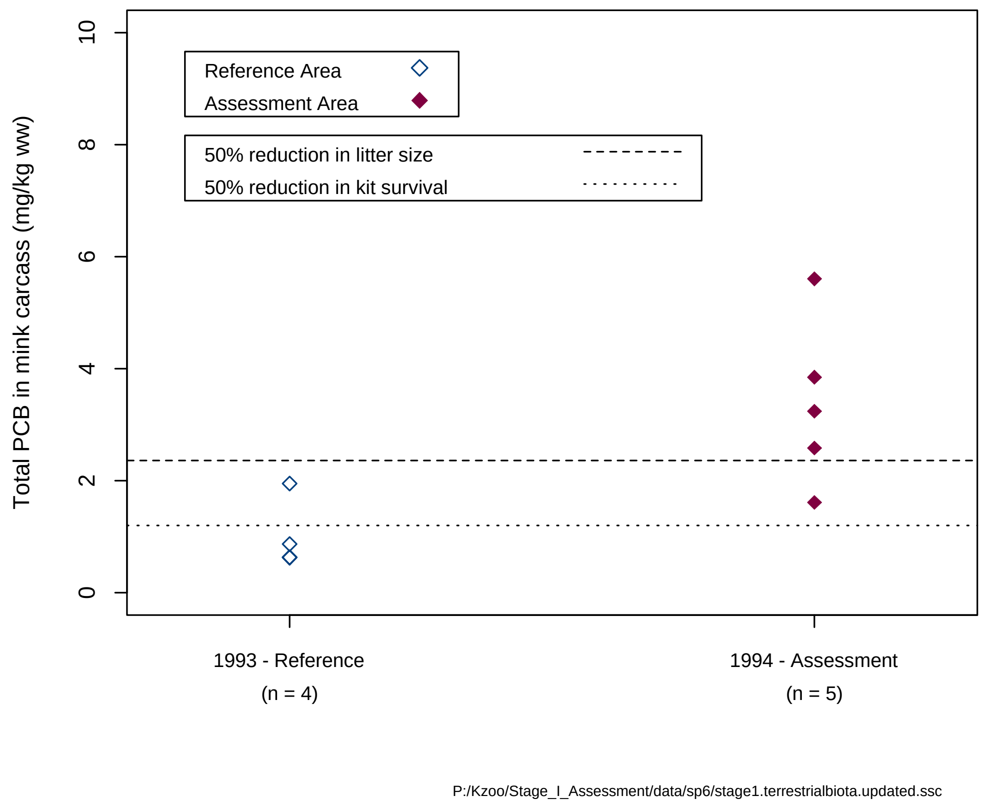


Figure 7.8. Total PCB concentrations in mink carcasses collected in reference and assessment locations. Two of the four reference samples collected in 1993 have concentrations of approximately 0.63 mg/kg ww and overlap in this figure.

Source: Camp Dresser & McKee, 1997.

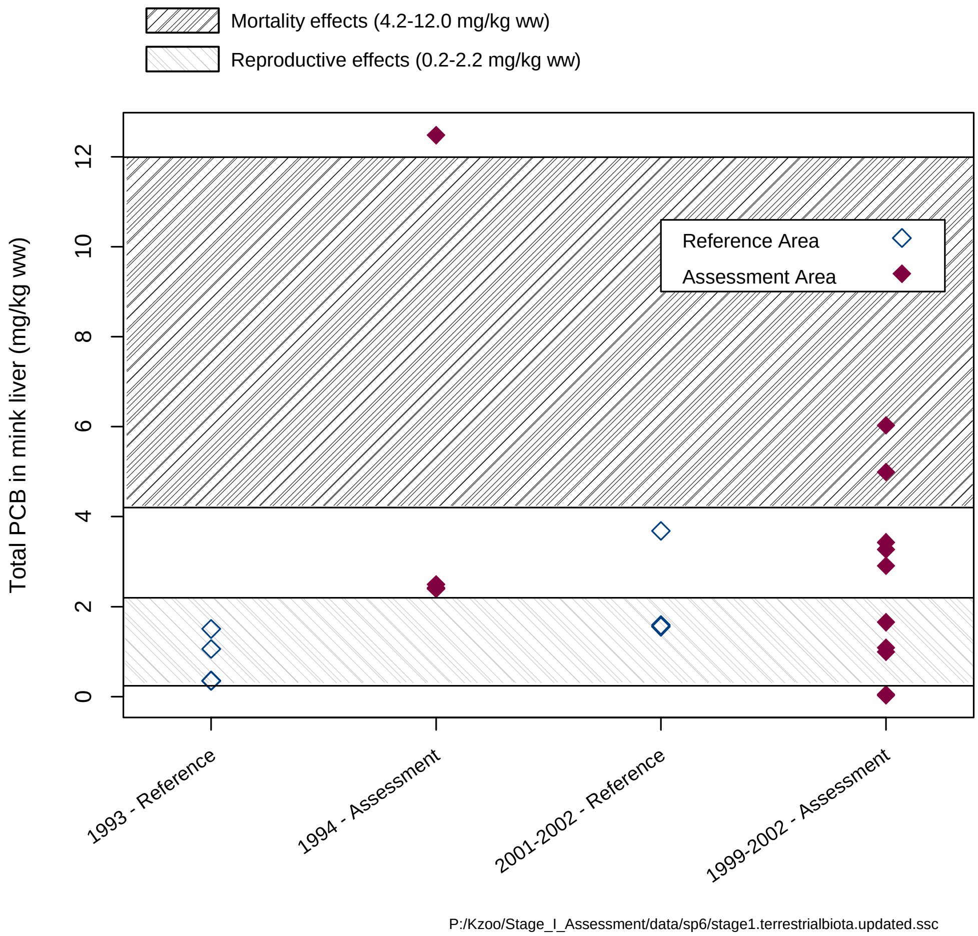


Figure 7.9. PCB concentrations in mink livers from reference and assessment locations compared to injury threshold ranges.

Sources: Camp Dresser & McKee, 1997; Michigan State University Aquatic Toxicology Laboratory, 2002b.

7.9 Total PCBs in Small Mammal, Shrew, and Muskrat Tissue

7.9.1 Data sources

The following data sources are used to evaluate injury to mice and muskrat in this section:

- PCB concentrations in mice collected in 1993 by Blasland, Bouck & Lee (2001)
- PCB concentrations in small mammals and shrews collected in 2000 by Michigan State University Aquatic Toxicology Laboratory (2002f; 2002g; 2002h; 2002i)
- PCB concentrations in muskrat livers collected in 1993 and 1994 by Camp Dresser & McKee (1997)
- PCB concentrations in whole body muskrat samples collected from 2000-2002 by Michigan State University Aquatic Toxicology Laboratory (2002c).

Blasland, Bouck & Lee (1994d) collected deer mice at one reference and four assessment locations along the Kalamazoo River in 1993. The reference location was upstream of Battle Creek. The assessment locations were near Plainwell Dam, Otsego Dam, Trowbridge Dam, and Lake Allegan Dam. Traps were set at 10 meter intervals across a grid at each of the sampling areas. The initial grids contained 48 traps each, and were extended spatially, with additional traps, until 10 mice were collected at each location. Whole body samples were analyzed (without gut contents) for PCB concentrations as Aroclors.

Michigan State University Aquatic Toxicology Laboratory (2002f, 2002g, 2002h, 2002i) collected a variety of small mammal species from a reference location in Fort Custer and an assessment location near Trowbridge Dam in 2000. A total of 34 deer mouse, eastern chipmunk, jumping mouse, red squirrel, and shrew (order Insectivora) samples were collected from Fort Custer. A total of 38 deer mouse, meadow vole and shrew samples were collected from near Trowbridge Dam.

In conjunction with the mink trapping study discussed in Section 7.8.1, Camp Dresser & McKee (1997) collected muskrat from one reference location (Battle Creek) and four assessment locations (near Plainwell Dam, Otsego Dam, Trowbridge Dam, and Lake Allegan Dam). Traps were set in riparian areas of the Kalamazoo River between December 1993 and April 1994. Six muskrat were collected at each sample location. Liver tissue samples were analyzed for total PCBs as Aroclors. One liver sample from Trowbridge Dam was lost; thus there are only five liver samples from this location.

Whole body muskrat samples were collected by Michigan State University Aquatic Toxicology Laboratory (2002c) from a reference location in Fort Custer and an assessment location in Trowbridge. Three samples were collected in Fort Custer in 2000, and one in 2002. Four samples were collected at Trowbridge Dam in 2000 and three in 2001. These samples were analyzed for total PCBs as the sum of congeners.

7.9.2 Toxicity reference value derivation

Many studies have been conducted on the toxicity of PCBs to mammals. Since mink are the most sensitive mammal tested to date, the TRV concentrations used for mink (see Section 7.8.2) are used to evaluate injury to mice and muskrat as well. If concentrations in deer mouse and muskrat tissue are higher than those expected to cause reproductive effects in mink, further evaluation will be warranted. If they are lower than the mink TRVs, it would be reasonable to conclude that there is no evidence of injury to these species. Thus, whole body TRVs of 1.2 mg/kg ww (reduced reproduction) and 2.36 mg/kg ww (survival of offspring) are compared to whole body PCB concentrations in mice and muskrats, and a reproductive effects TRV range of 0.2 to 2.2 mg/kg ww in liver is compared to muskrat liver PCB concentrations.

7.9.3 Results

PCB concentrations measured in deer mice collected in 1993 and in deer mice, eastern chipmunk, meadow voles, and shrews collected in 2000 are slightly higher in assessment areas than in the reference areas (Table 7.21). However, PCB concentrations in only seven shrew samples of the 77 total small mammal and shrew samples from locations downstream of PRPs are near those associated with reproductive effects in mink. Three of these samples exceed the TRV associated with reduced survival of mink offspring (2.36 mg/kg ww). The maximum concentration in a whole body shrew sample is 3.15 mg/kg ww. However, the sensitivity of shrews to PCB toxicity is unknown. The available data do not indicate that most small mammals are injured by exposure to PCBs, although injury to shrews (Insectivora) is somewhat uncertain.

Measured PCB concentrations are higher in muskrat livers in assessment areas than in the reference location (Table 7.22). PCBs were not detected in the six muskrat livers collected from the Battle Creek reference location, and concentrations range from 0.03 to 1.18 mg/kg ww in assessment area samples. A muskrat liver sample from near Trowbridge Dam has the highest concentration, 1.18 mg/kg ww. Mean concentrations in assessment area muskrat livers are within the range of concentrations expected to be associated with toxic effects in mink. No PCB toxicity tests on muskrat are available to assess muskrat sensitivity to PCBs, but the muskrat is a rodent, and rodents tend to be less sensitive to PCB toxicity than mink (Shore and Douben, 1994). Whole body muskrat samples, however, do not exceed whole body mink reproductive effects TRVs (Table 7.23). Therefore, the available data do not conclusively indicate that the PCB concentrations measured in muskrat tissue are sufficient to cause injury to the muskrat, and this is therefore an area of uncertainty.

Table 7.21. Total PCB concentrations in small mammals and shrews

Location	Species	Number of samples	Total PCB concentration in whole body (mg/kg ww)[a]	Maximum exceeds mink reproductive effects TRV?[b]	Sample Year[c]
Battle Creek (reference)	Deer mouse	10	0.01 (0.01-0.04)	No	1993
Fort Custer (reference)	Deer mouse	9	0.01 (0.00-0.03)	No	2000
Fort Custer (reference)	Eastern chipmunk	5	0.00 (0.00-0.01)	No	2000
Fort Custer (reference)	Jumping mouse	3	0.08 (0.01-0.18)	No	2000
Fort Custer (reference)	Red squirrel	1	0.00	No	2000
Fort Custer (reference)	Shrew	16	0.01 (0.00-0.02)	No	2000
Upstream of Plainwell Dam	Deer mouse	10	0.09 (0.01-0.28)	No	1993
Downstream of Otsego Dam	Deer mouse	10	0.26 (0.09-0.38)	No	1993
Upstream of Trowbridge Dam	Deer mouse	10	0.12 (0.01-0.45)	No	1993
Trowbridge	Deer mouse	11	0.17 (0.02-0.55)	No	2000
Trowbridge	Eastern Chipmunk	1	0.57	No	2000
Trowbridge	Meadow vole	9	0.04 (0.01-0.08)	No	2000
Trowbridge	Shrew	17	1.31 (0.03-3.15)	Yes	2000
Downstream of Lake Allegan Dam	Deer mouse	10	0.06 (0.01-0.35)	No	1993

a. Mean (range). One-half the detection limit was used to calculate the mean if a concentration was below the analytical detection limit.
b. Whole body mink TRVs are 1.2 mg/kg PCBs for reduced litter size and 2.36 for reduced kit survival (Section 7.8.2).
c. Samples collected in 1993 from Blasland, Bouck & Lee (2001). Samples collected in 2000 from Michigan State University Aquatic Toxicology Laboratory (2002f, 2002g, 2002h, 2002i).

Table 7.22. Total PCB concentrations in muskrat livers

Location	Number of samples	Total PCB in liver (mg/kg ww)[a]	Maximum falls within mink reproductive effects TRV range (0.2-2.2 mg/kg)?
Battle Creek (reference)	6	ND	No
Upstream of Plainwell Dam	6	0.30 (0.03-0.70)	Yes
Downstream of Otsego Dam	6	0.13 (0.03-0.27)	Yes
Upstream of Trowbridge Dam	5	0.44 (0.06-1.18)	Yes
Downstream of Lake Allegan Dam	6	0.33 (0.09-0.49)	Yes

a. Mean (range). ND = not detected at detection limit of 0.02 or 0.03 mg/kg ww. Concentrations on a wet weight basis were calculated using the reported dry weight basis concentrations and moisture content in samples.

Source: Camp Dresser & McKee, 1997.

Table 7.23. Total PCB concentrations in whole body muskrat samples collected between 2000 and 2002

Location	Number of samples	Total PCB (mg/kg ww)[a]	Maximum exceeds mink reproductive effects TRV (1.2 mg/kg)?
Fort Custer (reference)	4	0.01 (0.01-0.03)	No
Trowbridge Dam	7	0.07 (0.01-0.11)	No

a. Mean (range).

Source: Michigan State University Aquatic Toxicology Laboratory, 2002c.

7.10 Bioaccumulation of PCBs from Floodplain Soils

The assessment of injuries to KRE wildlife in the preceding sections is based primarily on PCB concentrations measured in biota tissue samples (fish, bird eggs, and mammal tissue). However, these data samples represent only a small fraction of the organisms exposed to PCBs in the KRE, and in only a few selected areas. To address the uncertainty associated with the limited nature of the available tissue data, this section presents an evaluation of potential wildlife injuries based on the much more extensive KRE floodplain soil PCB concentration data that are available. A bioaccumulation exposure model is used to estimate soil concentrations at which food chain exposure for selected wildlife species would cause adverse toxicological impacts, and the estimated soil effects PCB concentrations are compared to the available data on PCB concentrations in KRE soils.

This analysis not only expands the scope of the Stage I KRE wildlife injury assessment but it also can serve as the foundation for restoration scaling approaches that rely on estimates of spatial extent and degree of injury, such as Habitat Equivalency Analysis (HEA; NOAA, 1997). HEA is a tool that can be used to determine the type and extent of habitat restoration that is necessary to offset injuries to natural resources caused by hazardous substances. HEA balances the duration, degree, and spatial extent of habitat services lost because of the injuries with the duration, degree, and spatial extent of habitat services gained through habitat restoration. The spatially-based analysis presented in this section can be used to help define the floodplain habitat services lost because of the presence of PCBs in the soils and the potential for the PCBs to cause adverse effects on wildlife.

7.10.1 Data sources

The following data sources are used in this evaluation:

- Floodplain surface soil samples collected at depths of 0-6 in. by Blasland, Bouck & Lee (2001) in July and August 1993, and analyzed for PCBs as Aroclors. Methods are detailed in Blasland, Bouck & Lee (1994f).

- Floodplain surface soil samples collected at depths of 0-6 in. by Blasland, Bouck & Lee (2001) between November 1993 and February 1994, and analyzed for PCBs as Aroclors. Methods are detailed in Blasland, Bouck & Lee (1994c).

- Floodplain surface soil samples collected as part of a focused sampling program designed by MDEQ (Blasland, Bouck & Lee, 2000b, 2001). Samples were collected from April to August 2000 and analyzed for PCBs as Aroclors. The sample locations were selected by MDEQ to characterize known and suspected PCB point sources and wildlife habitat areas.

- Floodplain surface soil samples from the Plainwell and Otsego City impoundments collected at depths of 0-6 in. in the spring and summer of 2001 as part of the Stage I Removal Assessment by R.F. Weston for EPA and Camp Dresser & McKee (2002a) for MDEQ. Sampling was designed to supplement information provided by existing samples (U.S. EPA and MDEQ, 2002). Initial samples were collected on a grid across the former impoundments, and then additional samples were collected in clusters to better define the PCB distribution observed in grid samples.

Because terrestrial biota generally are exposed to surface soils, only surface samples from these studies were considered. The majority (91%) of these surface samples were collected at depths of 0-6 inches. Surface samples for which an entire 0-6 inch section was not available, or for which a deeper surface section (up to 14 inches) was collected, were also used in this analysis. Although soils at depths greater than 6 inches may become available to biota via burrowing or human disturbance, it is assumed that PCBs in this surface layer are the best estimate of what is biologically available.

Only soil samples collected from the Kalamazoo River floodplain or from one of the former impoundments are included in this analysis. Former impoundment and river boundaries were defined using historical and current aerial photographs (USGS, 1956, 1965; Air-Land Surveys, 1999). Samples were identified as being within one of the three former impoundments if they were located within these defined boundaries and were not located within the river channel. Samples were identified as being within the Otsego City impoundment if they were located upstream of the Otsego Dam and downstream of the former Plainwell Dam. Samples were identified as being within the Kalamazoo River floodplain if they were not located within one of the former impoundments or the Otsego City impoundment, and if they were identified as Kalamazoo River floodplain samples by Blasland, Bouck & Lee (2001). No other available sample data were included in the analysis of floodplain soil data.

7.10.2 Toxicity reference value derivation

There are no state or federal regulatory standards for the protection of biological resources from hazardous substance concentrations in soil. However, non-enforceable guidelines for PCB concentrations in soil that are protective of ecological receptors are available from several sources.

The DOI (as cited in U.S. EPA, 1990) has found that, generally, PCB concentrations in soils of less than 1-2 mg/kg dw are protective of wildlife. Additionally, preliminary remediation goals (PRGs) have been compiled by the U.S. Department of Energy (DOE) (Efroymson, et al., 1997). These PRGs are concentrations that are anticipated to protect ecological endpoints and were extracted largely from toxicological benchmarks developed for the Oak Ridge National Laboratory (ORNL). The PRG for PCBs in soil, 0.371 mg/kg dw, is not intended to be protective of a specific species, but is the lowest value of PRGs developed for wildlife, plants, and soil invertebrates.

The food web model that was developed for the KRE by the MDEQ in its ecological risk assessment (Camp Dresser & McKee, 2003b) was used to calculate soil concentrations that would result in wildlife exposure at dietary no effect and low effect concentrations. The MDEQ ERA developed threshold concentrations in soils for the American robin, white-footed and deer mouse, great horned owl, and red fox. The food web model incorporates assumptions regarding the dietary composition of these species and the uptake of PCBs from soil into their food items, which are described in detail by Camp Dresser & McKee (2003b). Table 7.24 presents the soil PCB concentrations that are estimated to produce exposure for the selected wildlife species at

Table 7.24. Site-specific soil PCB threshold concentrations for evaluating injury to selected wildlife species

Receptor species	No effect TRV-based soil threshold (mg/kg dw)	Low effect TRV-based soil threshold (mg/kg dw)
Robin	6.5	8.1
Mouse	21	63
Owl	2.9	8.5
Fox	5.9	29.5

their dietary no effect and low effect values (Camp Dresser & McKee, 2003b). The Trustees use the low effect TRV based soil thresholds as developed by MDEQ to evaluate injury to floodplain soils in this Stage I Assessment.

7.10.3 Results

The combined data sources described in Section 7.10.1 yield 475 surface soil samples from within the KRE floodplain or former impoundment areas. Of these, 222 are located within the boundaries of one of the former impoundments (Plainwell, Otsego, Trowbridge), and 90 are located in the floodplain area of the Otsego City impoundment. An additional 163 samples are located within the floodplain of the Kalamazoo River, but not within the boundaries of any of the four former or current impoundments. Analytical detection limits for the floodplain soil samples included in this discussion range from 0.037 to 0.35 mg/kg.

Former Plainwell impoundment

Surface soil PCB concentrations in the former Plainwell impoundment range from 0.053 to 134 mg/kg. Concentrations in 53% of the samples from within the former impoundment boundary are greater than the injury threshold for robins of 8.1 mg/kg, and 3% are more than 10 times greater (Figure 7.10). Concentrations in 53% of the surface soil samples also exceed the injury threshold for owls of 8.5 mg/kg, and 2% are more than 10 times greater (Figure 7.11). Concentrations in 26% of the surface soil samples from the Plainwell impoundment exceed the fox injury threshold (Figure 7.12). Five percent of the samples have concentrations greater than the mouse threshold (Figure 7.13).

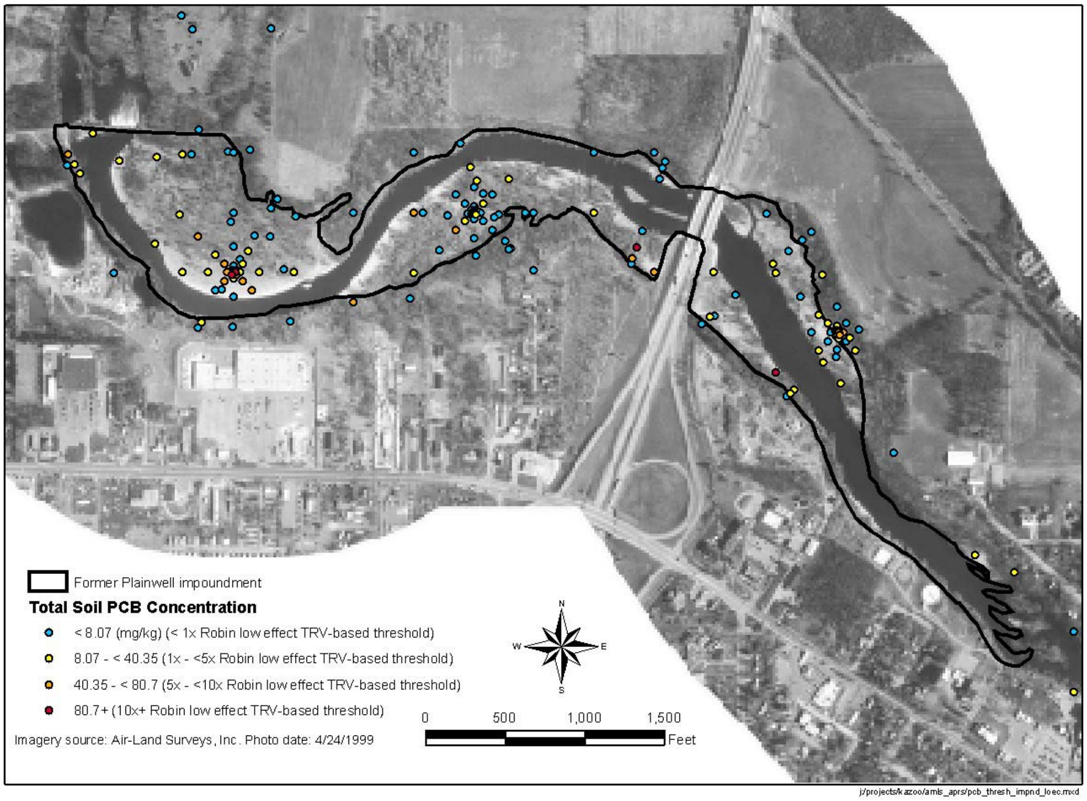


Figure 7.10. Surface soil PCB concentrations in the former Plainwell impoundment compared to the soil threshold for injury to robins. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

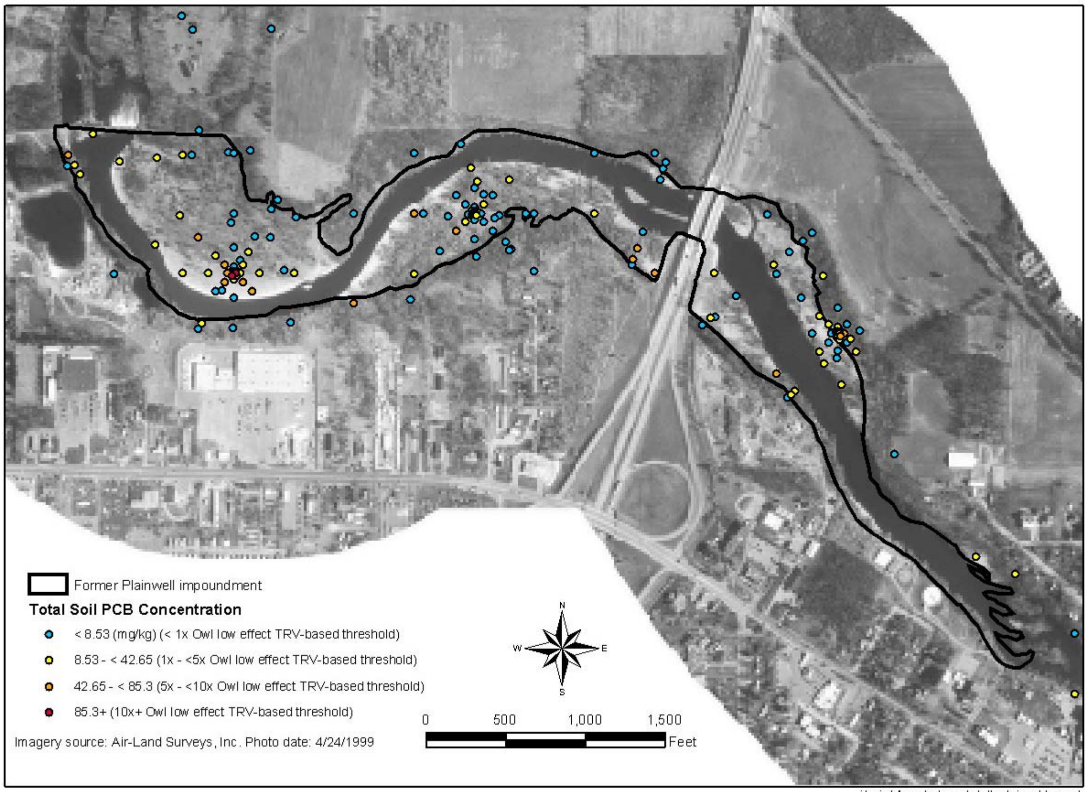


Figure 7.11. Surface soil PCB concentrations in the former Plainwell impoundment compared to the soil threshold for injury to owls. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

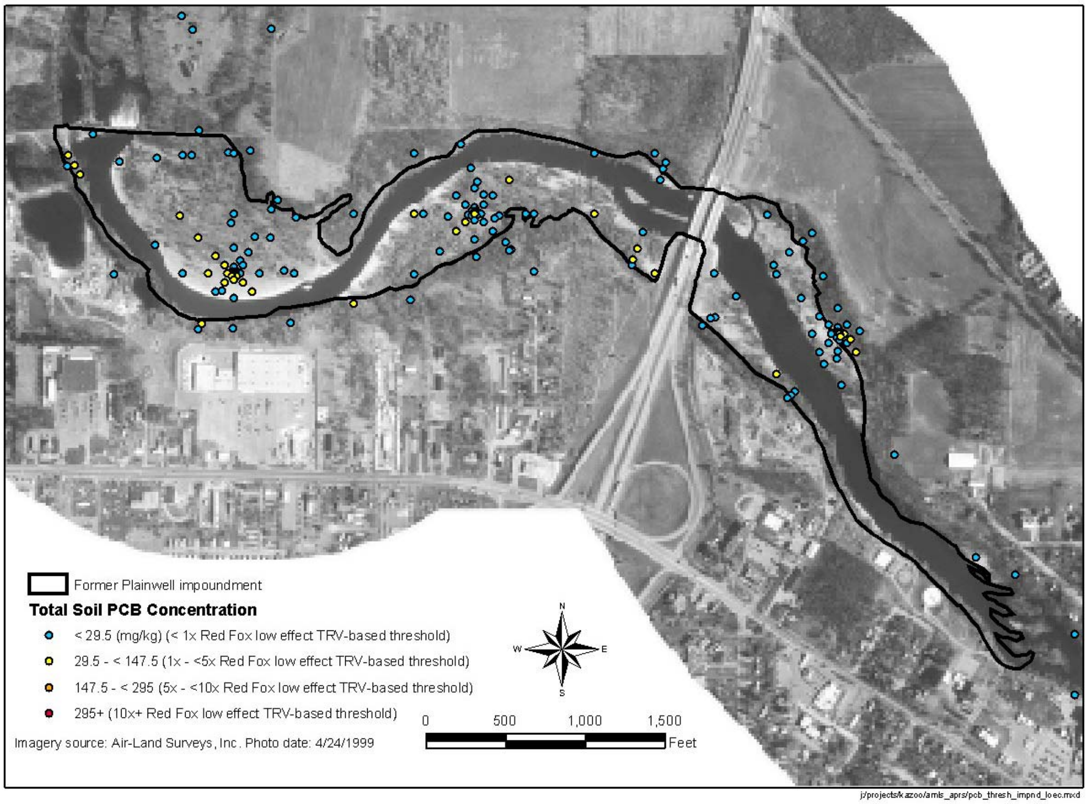


Figure 7.12. Surface soil PCB concentrations in the former Plainwell impoundment compared to the soil threshold for injury to foxes. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

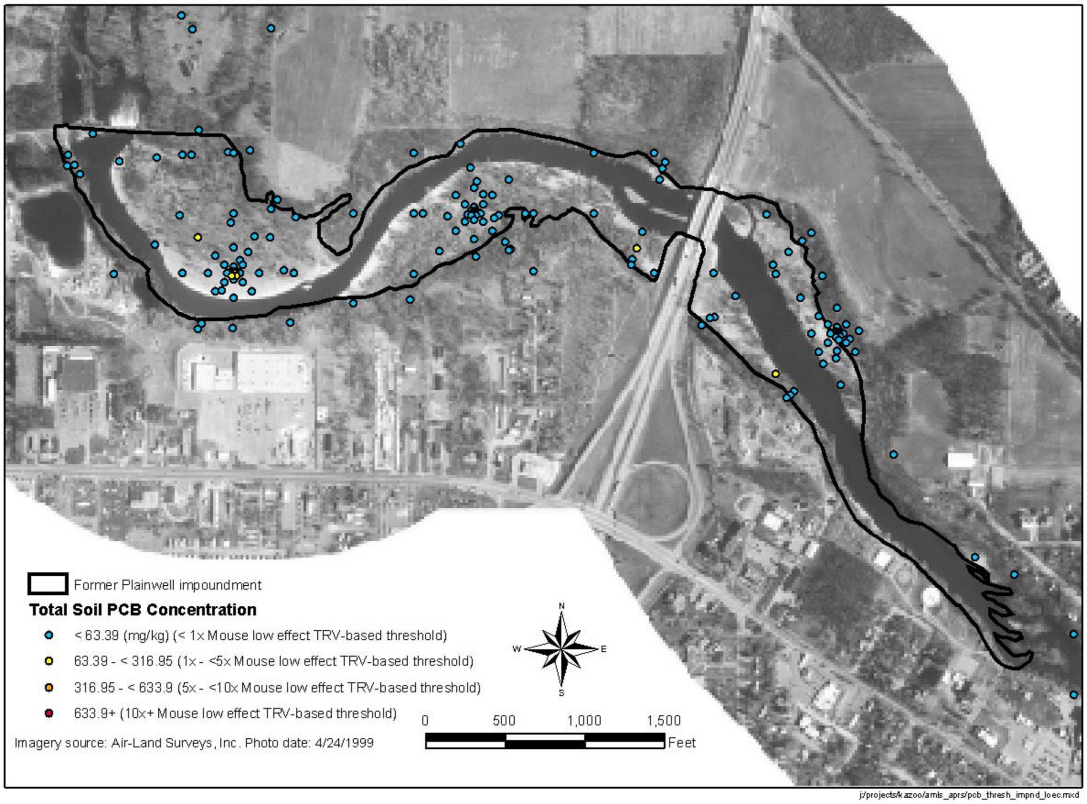


Figure 7.13. Surface soil PCB concentrations in the former Plainwell impoundment compared to the soil threshold for injury to mice. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

Otsego City impoundment

PCB concentrations in surface soil from the Otsego City impoundment area are lower than those observed just upstream in the former Plainwell impoundment. Forty-two percent of the 90 samples from the Otsego City impoundment area have detectable concentrations of PCBs, and those concentrations range from 0.0034 to 58.9 mg/kg. Eight percent of samples exceed the injury threshold for robins, and four percent exceed the injury threshold for owl, although only one sample is more than five times greater than either threshold (Figures 7.14 and 7.15). The sample with the highest concentration, located just upstream of the dam, is the only case where surface soil concentrations in the Otsego City impoundment exceed the injury threshold for red fox (not plotted). None of the surface soil samples collected from around this impoundment have concentrations that are greater than the mouse threshold (not plotted).

Former Otsego impoundment

All but two surface soil samples of the 26 collected from within the former Otsego impoundment have detectable levels of PCBs. Detected concentrations range from 0.13 to 61 mg/kg. Fifty-eight percent of samples exceed the injury thresholds for both robin and owl; only one sample is more than five times greater than either threshold (Figures 7.16 and 7.17). Fifteen percent of samples also exceed the red fox injury thresholds (Figure 7.18). No samples exceed the injury threshold for mouse (not plotted).

Former Trowbridge impoundment

All of the 63 surface soil samples collected from the former Trowbridge impoundment have detectable levels of PCBs. Concentrations range from 0.051 to 81.1 mg/kg. Fifty-four percent of samples from within the impoundment boundary exceed the injury threshold for robins, with 2% more than 10 times greater than this threshold (Figure 7.19). Fifty-two percent of samples have PCB concentrations greater than the owl injury threshold (Figure 7.20), 21% exceed the fox threshold (Figure 7.21), and two samples exceed the mouse threshold (Figure 7.22).

Other Kalamazoo River floodplain soils

Soil samples collected from the Kalamazoo River floodplain outside of impoundment or former impoundment areas generally have lower PCB concentrations than those that were collected in the three former impoundments and in the Otsego City impoundment. Fifty of the 163 surface samples from the floodplain do not have detectable levels of PCBs. Detected concentrations range from 0.026 to 29.3 mg/kg. Two of the surface samples in the Kalamazoo River floodplain are higher than the injury thresholds for robins and owls, both by less than 5 times. No samples exceed the mouse or fox injury thresholds.

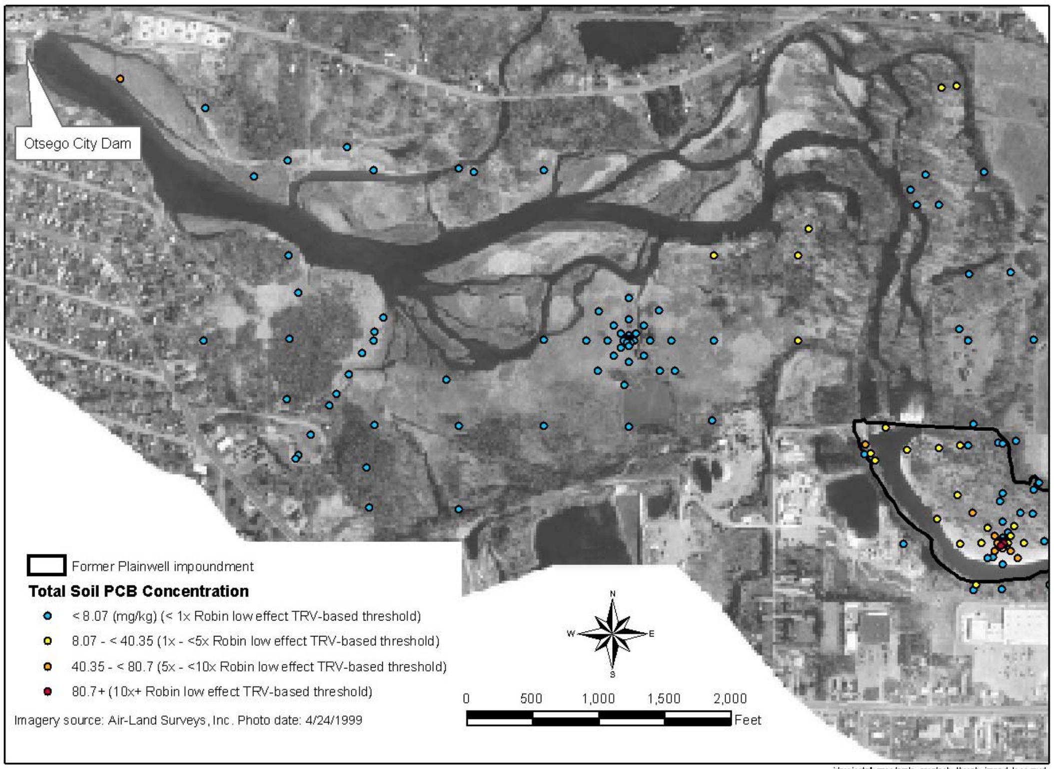


Figure 7.14. Surface soil PCB concentrations in the Otsego City impoundment compared to the soil threshold for injury to robins. No boundary is available for the Otsego City impoundment. Samples located within or upstream of the former Plainwell impoundment boundary in this figure are not included in the description of data in Section 8.3.3.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

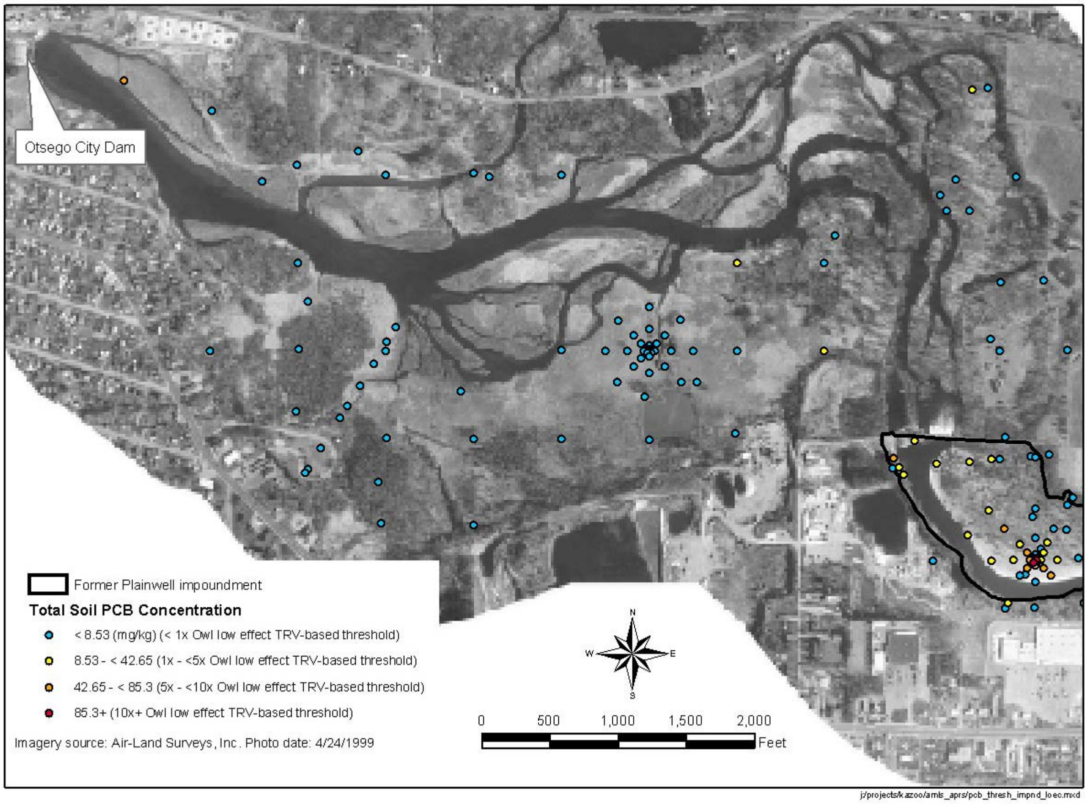


Figure 7.15. Surface soil PCB concentrations in the Otsego City impoundment compared to the soil threshold for injury to owls. No boundary is available for the Otsego City impoundment. Samples located within or upstream of the former Plainwell impoundment boundary in this figure are not included in the description of data in Section 8.3.3.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

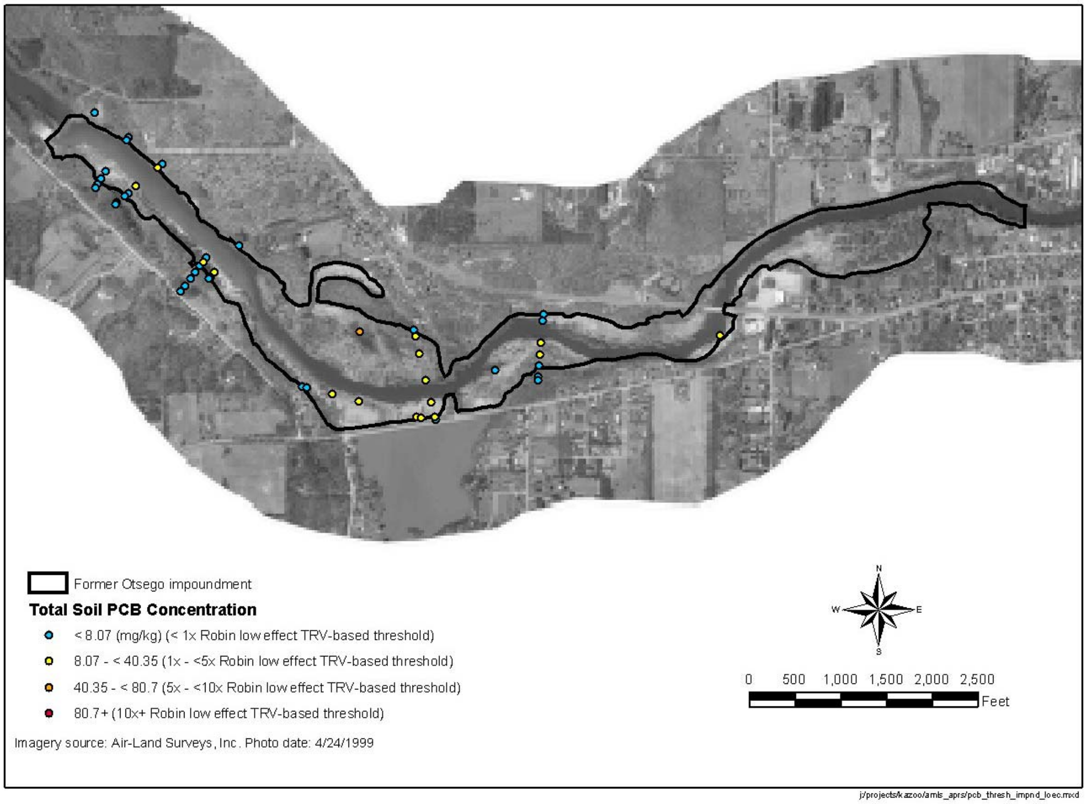


Figure 7.16. Surface soil PCB concentrations in the former Otsego impoundment compared to the soil threshold for injury to robins. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

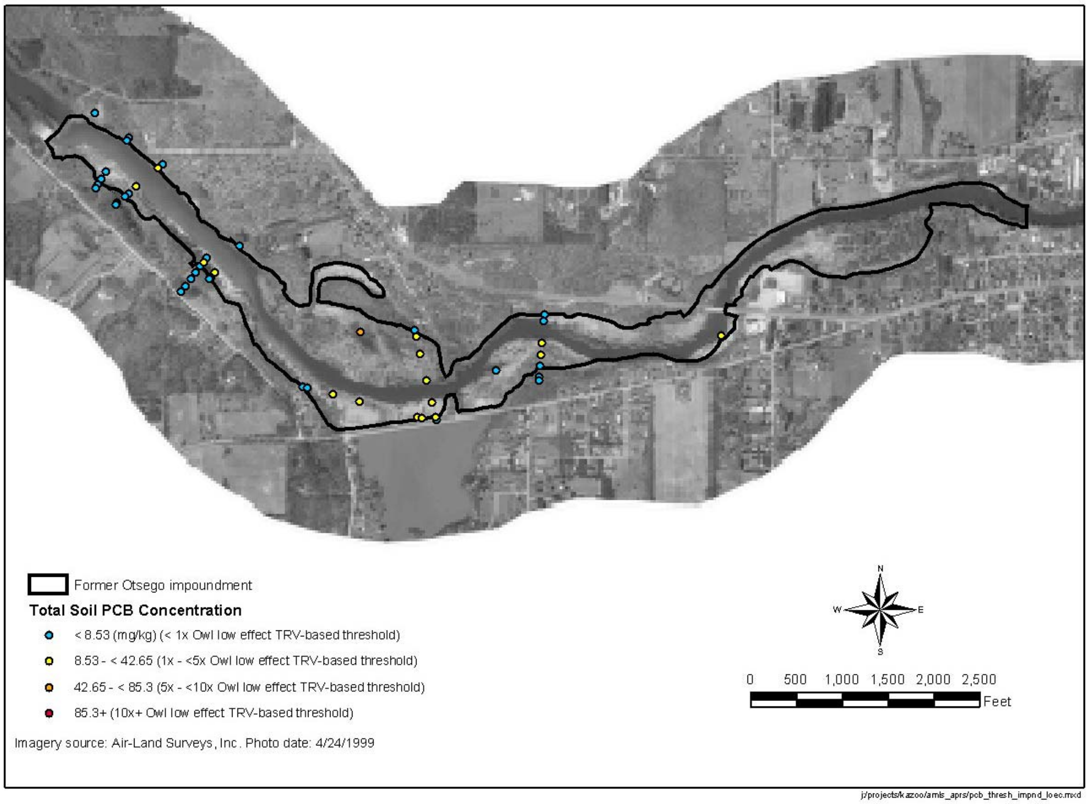


Figure 7.17. Surface soil PCB concentrations in the former Otsego impoundment compared to the soil threshold for injury to owls. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

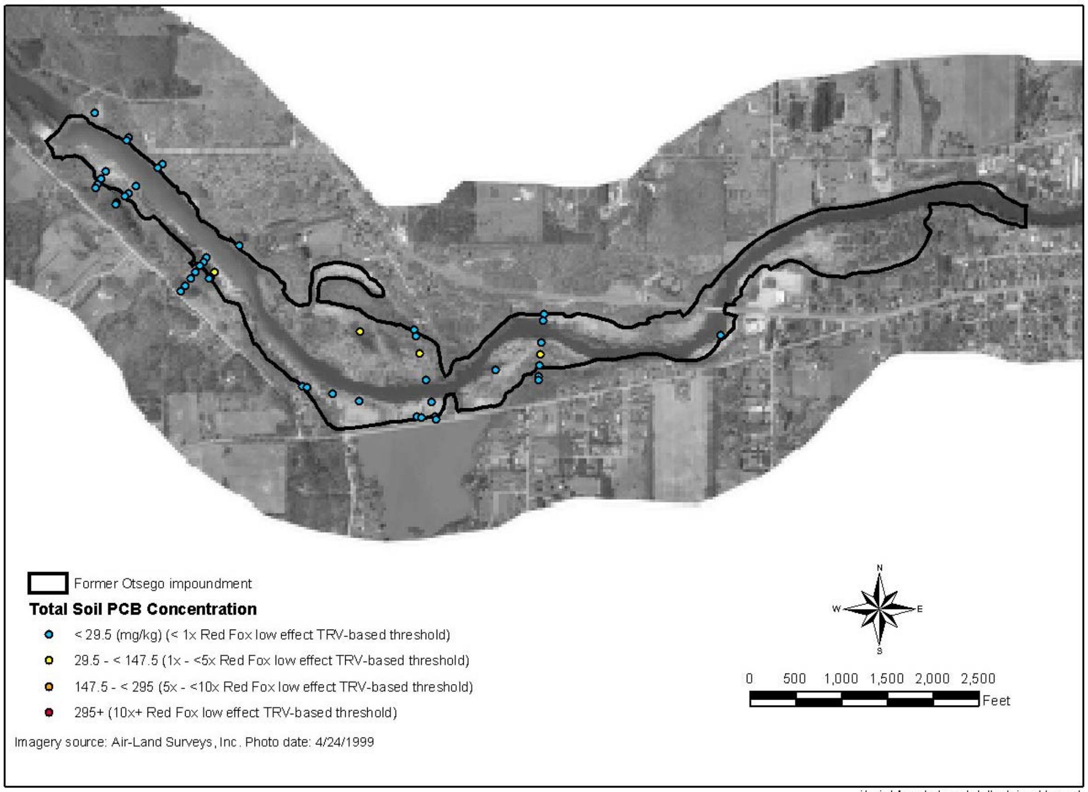
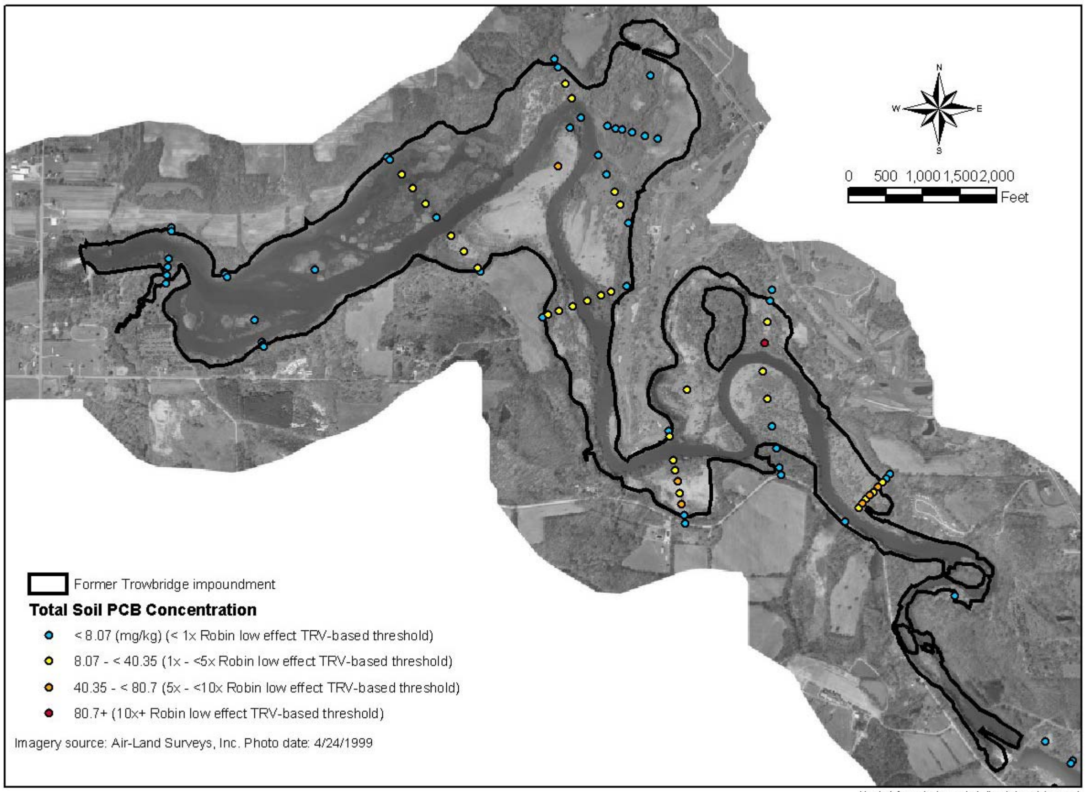


Figure 7.18. Surface soil PCB concentrations in the former Otsego impoundment compared to the soil threshold for injury to foxes. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.



j:/projects/kazoo/amls_aprs/pcb_thresh_impnd_loec.mxd

Figure 7.19. Surface soil PCB concentrations in the former Trowbridge impoundment compared to the soil threshold for injury to robins. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

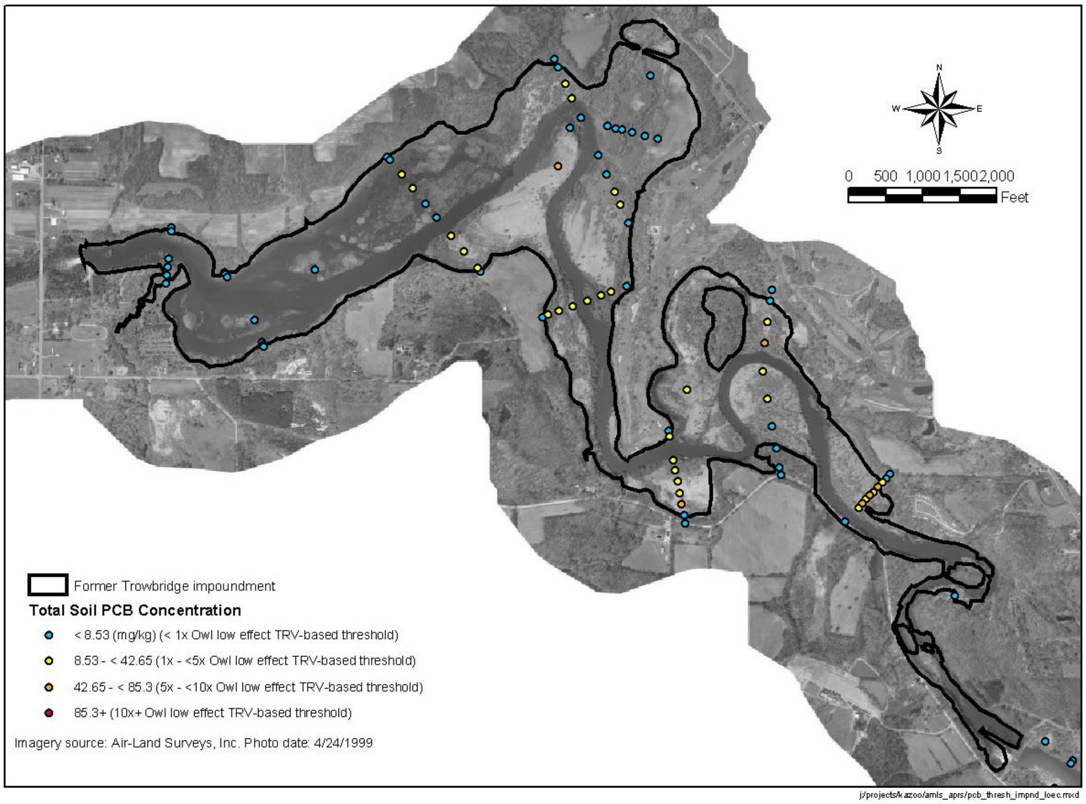


Figure 7.20. Surface soil PCB concentrations in the former Trowbridge impoundment compared to the soil threshold for injury to owls. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

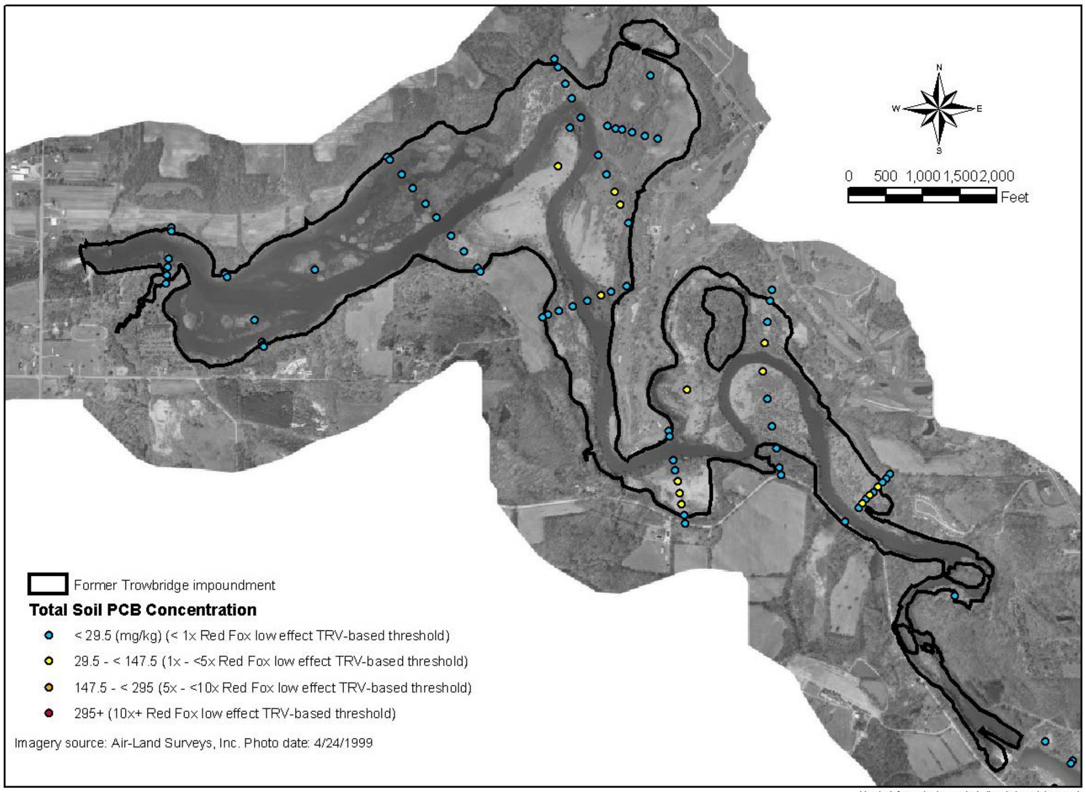
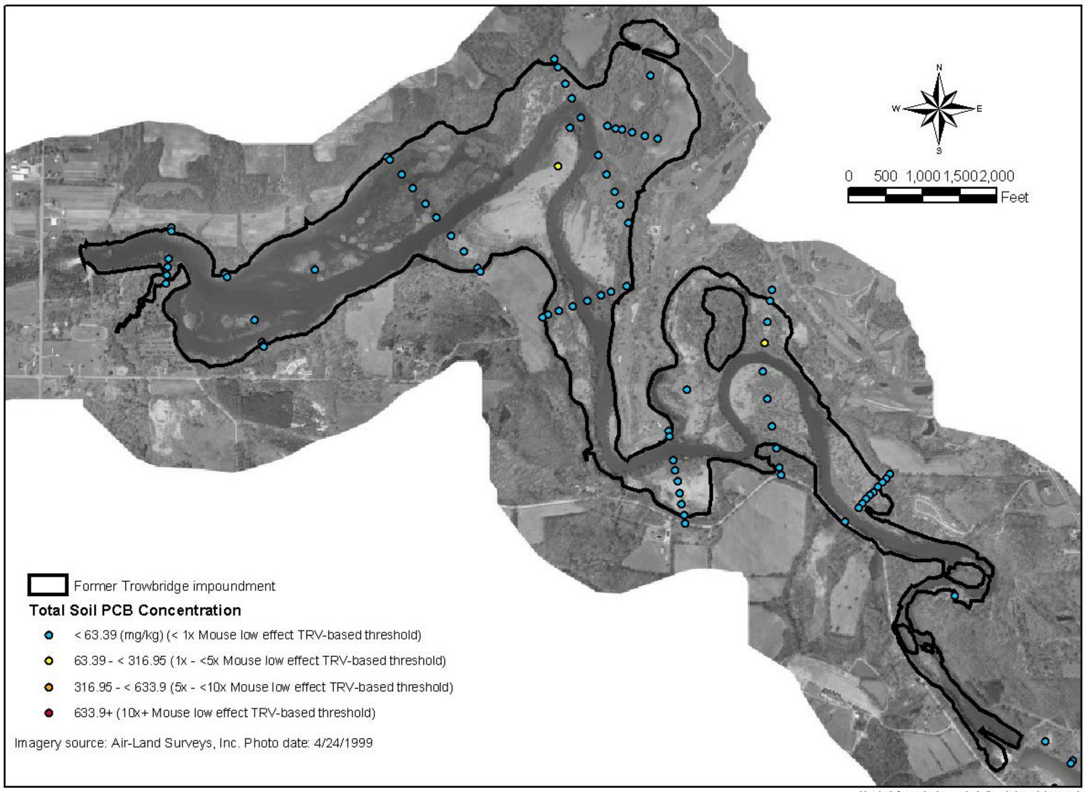


Figure 7.21. Surface soil PCB concentrations in the former Trowbridge impoundment compared to the soil threshold for injury to foxes. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.



j:/projects/kazoo/amls_aprs/pcb_thresh_impnd_loec.mxd

Figure 7.22. Surface soil PCB concentrations in the former Trowbridge impoundment compared to the soil threshold for injury to mice. Points that are located beyond the boundary of the impoundment are not included in analysis, but are plotted for reference.

Sources: Blasland, Bouck & Lee, 2001; Camp Dresser & McKee, 2002a, 2003b.

7.10.4 Conclusions of the floodplain soil evaluation

An evaluation of floodplain surface soil PCB concentrations shows that many floodplain areas of the KRE contain PCBs that exceed concentrations predicted to cause adverse effects to wildlife via food chain exposure. In particular, many of the samples from the former impoundments (Plainwell, Otsego, and Trowbridge) contain surface soil PCB concentrations that exceed thresholds for robins and great horned owls. In many cases, the soil concentrations are many times greater than the thresholds. However, soil thresholds for mice and red fox are only rarely exceeded. This analysis indicates that the spatial extent of injuries to some KRE wildlife species may be quite large, covering many acres of the former impoundments and other floodplain areas.

In the preceding sections of this chapter, PCB concentration data for KRE robin and great horned owl eggs and mice tissue were presented and evaluated. The results of this floodplain soils evaluation are consistent with the evaluation of PCB concentration data for great horned owl and mice. However, PCB concentrations measured in two robin eggs from the former Plainwell impoundment are less than TRVs for egg mortality (Section 7.6.3), while this evaluation of soil PCB concentrations indicates that robins are exposed to PCB concentrations in soils expected to cause adverse effects. The differences between the conclusions from the tissue data evaluation and from the floodplain soil data analysis may have several possible causes:

- The tissue collection data may not be as representative of PCB exposure as are the soil data. The tissue sample sizes were fairly small relative to the soil data, and the exposure history of the collected organisms is unknown.

- The bioaccumulation and dietary exposure models used in the MDEQ ERA to derive soil thresholds may not accurately reflect actual exposure. The bioaccumulation model predicts higher exposure for robins than indicated by the available tissue data. Field data collections are currently being conducted to more fully characterize PCB bioaccumulation in the KRE (Giesy Ecotoxicology, 2001), and the results of these studies may help explain the apparent discrepancies between the currently available field data and the bioaccumulation model.

- A combination of the two factors.

Nevertheless, the analysis of the available floodplain soil PCB data indicates that wildlife species are exposed to PCBs in KRE floodplain areas at concentrations sufficient to cause adverse effects over fairly large areas.

7.11 Conclusions

This chapter presents a Stage I Injury Assessment for KRE wildlife based on available information and data. Two types of injury to wildlife are evaluated: exceedences of the FDA tolerance level for PCBs in waterfowl tissue; and adverse changes in viability resulting from toxicological effects of PCBs.

Available data on PCB concentrations in mallard duck tissue indicate that injury has occurred. The PCB concentration in one of two available mallard duck samples was 6 times higher than the FDA tolerance level. Thus the Trustees conclude that mallard ducks are injured according to the definition in 43 C.F.R. § 11.62 (f)(1)(ii). However, because the data are minimal, the Stage I Assessment does not draw any conclusions about the spatial or temporal extent of this injury.

Table 7.25 summarizes the conclusions of the Stage I Injury Assessment about injuries to KRE wildlife based on adverse changes in viability. The Trustees conclude that bald eagles, piscivorous birds, predatory birds, and mink are injured by exposure to KRE PCBs. Additionally, omnivorous birds (such as robin) are probably exposed to PCBs in KRE floodplain surface soils at concentrations sufficient to cause injury. The severity of the injuries probably ranges from sublethal effects (e.g., reduced chick growth rate, embryo abnormalities) in some piscivorous birds to substantially reduced reproductive success in bald eagles and mink. Based on the available information, the injuries to wildlife most likely occur throughout the KRE, and they have been occurring for several decades (probably beginning soon after the PCB releases into the KRE began).

KRE bald eagle data support the conclusion that eagles are injured by exposure to PCBs. Bald eagle reproductive rates are lower near the Kalamazoo River than in other coastal and inland Michigan sites. Although nests have been attempted since 1990, the first offspring were not successfully reared until 1998, and the average nesting success between 1990 and 2003 was only 0.2 young per nest, far below normal replacement rates. Bald eagle eggs and plasma collected in the KRE have total PCB concentrations sufficient to substantially reduce reproductive rates. Furthermore, PCB concentrations in fish, an important bald eagle food source, exceed dietary TRVs for toxicological effects. The ERA conducted by the MDEQ also concluded that bald eagles are at high risk from exposure to PCBs. Therefore, the Trustees conclude that KRE bald eagles are injured by exposure to PCBs, and that the injury is increased egg mortality and decreased reproductive success.

Table 7.25. Summary of Stage I Assessment conclusions regarding toxicological injuries to wildlife

Species/resource	Injured?[a]	Nature of injury	Basis for conclusions	Probable spatial extent of injury
Bald eagle	Yes	Egg mortality, reduced productivity	Fish [PCB] > dietary TRVs Productivity data Egg [PCB] > TRVs Plasma [PCB] > TRVs	Two breeding areas in Allegan State Game Area, and one near New Richmond. Depending on suitability of habitat, rest of river downstream of PRP facilities.
Piscivorous birds	Yes	Sublethal effects Reproductive effects in sensitive species Egg mortality	Fish [PCB] > dietary TRVs MDEQ ERA results Egg [PCB] > TRVs (e.g., great blue heron)	Entire river downstream of PRP facilities (based on fish [PCB]).
Predatory birds (e.g., great horned owl, red-tailed hawk)	Yes	Egg mortality Reproductive impairment	Egg [PCB] > TRVs MDEQ ERA results for great horned owl Floodplain soil PCB concentrations > great horned owl toxicity thresholds (some > 10 times)	Floodplains soils in former impoundments and other areas (based on exceedences of great horned owl thresholds). Egg data available only for Allegan State Game Area.
Passerine birds (e.g., robin, red-winged blackbird, wood thrush, yellow warbler) and wood duck	Possible	Reproductive impairment	Egg [PCB] ≅ sensitive species TRVs MDEQ ERA results for robins Floodplain soil PCB concentrations > robin toxicity thresholds (some > 10 times)	Floodplains soils in former impoundments and other areas (based on exceedences of robin thresholds).
Waterfowl (e.g., mute swan)	No	-	Egg [PCB] < TRVs	-
Mink	Yes	Reproductive impairment	Fish [PCB] > dietary TRVs Mink whole body and liver [PCB] > TRVs MDEQ ERA results Anecdotal trapping success data	Entire river downstream of PRP facilities.

Table 7.25. Summary of Stage I Assessment conclusions regarding toxicological injuries to wildlife (cont.)

Species/resource	Injured?[a]	Nature of injury	Basis for conclusions	Probable spatial extent of injury
Red fox	No	Reproductive impairment	MDEQ ERA concludes risk for red fox unlikely unless diet consists of prey which have substantial amounts of PCBs Floodplain soil PCB concentrations > red fox toxicity thresholds	Exceedences occur within limited areas within floodplains soils and former impoundments and other areas.
White-footed/deer mouse	No	-	Mouse whole-body [PCB] < TRVs Floodplain soil PCB concentrations < toxicity thresholds (few exceedences) MDEQ ERA results	-
Muskrat	No	-	Muskrat liver [PCB] < TRVs Muskrat whole-body [PCB] < TRVs MDEQ ERA results	-

a. Yes = more likely than not that PCB exposure is at least a contributing factor to adverse changes in organism viability. Possible = some evidence for injuries, but important uncertainties exist. No = evidence for no injury.

Other piscivorous bird species are also exposed to elevated dietary PCB concentrations. Many of the PCB concentration measurements in whole body fish from the KRE exceed dietary toxicological benchmarks for causing sublethal effects in birds, and some exceed concentrations shown to cause embryomortality in species that are sensitive to PCBs. PCBs have also been measured in KRE great blue heron eggs at concentrations greater than thresholds for reduced hatching success. Although the PCB sensitivity of KRE piscivorous bird species is unknown, the available data support the conclusion that birds that consume fish from the KRE, such as the great blue heron or belted kingfisher, are injured from exposure to PCBs. Based on the spatial extent of PCB contamination in KRE fish, the injury is expected to occur throughout the Kalamazoo River and Portage Creek downstream of PRP facilities.

There are limited data on the PCB exposure of predatory birds in the KRE. PCB concentrations measured in great horned owl and red-tailed hawk eggs fall within the range expected to cause reduced egg hatching success. Additionally, the concentration of TCDD-eqs from PCBs in great horned owl eggs falls within the range associated with embryomortality effects in sensitive species. Although the data are limited, they support the conclusion that predatory birds are exposed to PCBs at concentrations sufficient to cause toxicological effects. The ERA conducted by the MDEQ also concluded that great horned owls are at high risk from exposure to PCBs. Therefore, the Trustees conclude that at least some predatory birds of the KRE are injured.

Egg PCB concentration data for many species of passerine birds (robin, red-winged blackbird, tree swallow, wood thrush, yellow warbler) and waterfowl (great blue heron, mute swan and wood duck) are less than threshold ranges for toxicological effects for tolerant species, but are within or greater than ranges for sensitive species. Egg PCB concentrations in eastern bluebird and house wren eggs also fall within the range for embryomortality in more tolerant species. The sensitivities of the species collected are not known. Therefore, these data indicate that these bird species may be injured by their exposure to KRE PCBs, depending on their sensitivity to PCB toxicity. In contrast, an evaluation of PCB concentrations in floodplain surface soils using the soil threshold concentrations developed in the MDEQ ERA food web model predicts that omnivorous birds such as robins are exposed to PCBs at concentrations sufficient to cause reproductive impairment. Soils in the former impoundments and other floodplain areas contain PCBs that are many times higher than soil thresholds for food chain effects to robins. The discrepancy between the predictions of the bioaccumulation model and the observed robin egg PCB data may be caused by a lack of representativeness of the available robin egg data, inaccuracies in the model, or a combination of both. Therefore, the Trustees conclude in this Stage I Injury Assessment that passerine species are possibly injured by exposure to PCBs.

Several types of data support the conclusion that KRE mink are injured by exposure to PCBs. PCB concentrations in fish from throughout the KRE exceed dietary thresholds for adverse reproductive effects in mink. Measured PCB concentrations in mink carcasses and livers captured from the KRE exceed thresholds for the same effects. Anecdotal trapping information suggests that mink populations along the Kalamazoo River downstream of PRP facilities are suppressed. The MDEQ ERA concludes that mink are at high risk from PCB exposure. Therefore, the Trustees conclude that mink are injured by exposure to KRE PCBs.

Available data indicate that KRE red fox are unlikely to be injured. The MDEQ ERA also concluded red fox are at low risk from PCB exposure, unless foraging is concentrated in riparian areas and the diet consists largely of organisms that have accumulated high concentrations of PCBs. Floodplain soils in some areas of the KRE contain PCBs at concentrations predicted to cause toxicity (based on the MDEQ ERA), but exceedences are infrequent.

Available data indicate that muskrat and mice are not injured. Measured PCB concentrations in KRE muskrat livers and whole-body samples are less than TRVs for toxic effects. The MDEQ ERA also concluded that these organisms are at low risk from PCB exposure.

Finally, the wildlife injury conclusions of the Stage I Assessment are limited to the species, resources, and areas for which relevant data are available. Bald eagles and mink are two key species in the assessment, and ample injury data and information are available for these species. However, PCB exposure data are available for only a small subset of the bird and mammal species present in the KRE, and only the bald eagle has been studied for information on actual adverse effects in the field. Therefore, the extent of injuries to wildlife species in the KRE may be greater than indicated by the available data.